Patterns in reef fish assemblages as determined by baited remote underwater video (BRUV) along the western side of False Bay: effects of site, depth, and protection status

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Abstract

By protecting ecosystems from exploitation, no-take zones are considered the principal means by which marine species and their populations can be conserved for future generations. To be successful, no-take zones require continuous monitoring of the fish community to evaluate the response of marine ecosystems to anthropogenic impacts and environmental change. Obtaining an understanding of the patterns of species composition, abundance, and distribution, allows monitoring efforts to be focused, efficient, and properly interpreted. Baited remote underwater video (BRUV) was used to examine the effects of site, depth, and level of protection, on the diversity and relative abundance of temperate reef fish within the Table Mountain National Park (TMNP) Marine Protected Area (MPA). Four no-take zones and adjacent exploited areas, subject to conventional management restrictions, were sampled monthly over a four-month period. A total of 36 species from three marine classes and 18 families was recorded. Species diversity (Shannon-Wiener index) was found to increase with sites closest to the mouth of the bay, whilst species abundance was found to increase with depth. Results indicated no consistent response to protection status among the sites at either the community or individual species level. However, the oldest no-take zone proclaimed for the purposes of reef conservation was found to harbour higher species diversity and a higher relative abundance of fish compared to its respective exploited area. Furthermore, the similar frequencies in which hottentot (*Pachymetopon blochii*) and roman (*Chrysoblephus laticeps*) were observed across the four study sites, suggests that these two commercially-important species are successfully recruiting inside and outside the no-take zones. These results indicate that physical factors, rather than protection status, within False Bay influence patterns of fish assemblage composition, abundance, and distribution. In future, and to improve comparability, assessments within the TMNP MPA should be designed to target similar locations and depth ranges within the bay. The success of no-take zones must be evaluated according to their individual design and management goals.

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CHAPTER 1

Literature Review

Overfishing is a pervasive threat to marine ecosystems (Jackson et al. 2001; Berkes et al. 2006; Molloy et al. 2009). Unsustainable exploitation rates jeopardise the future of human livelihoods and resources, as well as the integrity and resilience of marine ecosystems (Lotze et al. 2006). With more than 85% of fish stocks currently either fully exploited, overexploited, depleted or recovering (FAO, SOFIA report 2010), global fisheries are facing an unprecedented crisis that will only worsen as the human population continues to grow exponentially.

The use of marine protected areas (MPAs) is now widely advocated as one of the most pragmatic and tangible strategies with which to reverse this trend (Carr 2000; Hyrenbach et al. 2000). By setting aside areas in which human activities are reduced or eliminated, MPAs have the capacity to protect ecosystems from human disturbance and can address fisheries and conservation management objectives (Lester et al. 2009; Worm et al. 2009). However, despite their promise as a key component of a holistic ecosystem-based management approach, the historical pattern of *ad hoc* MPA design, limited enforcement and varying evaluation methods has produced contradicting evidence of their ecological effects (Carr 2000). In the face of mounting fishing pressure and an expanding global network of MPAs, there is a pressing need to monitor the performance of this management strategy to improve the effectiveness of management efforts, and optimise the allocation of human and financial resources (Carr 2000; Pomeroy et al. 2005).

Marine protected areas

Since traditional management approaches, such as bag and size limits, have failed to maintain the productivity of the oceans, MPAs are increasingly being implemented to protect biodiversity as well as the populations of commercially-important species and their habitats (Bohnsack et al. 2004; Kerwath 2005; WWF 2011). By zoning for different activities, MPAs promote a multi-purpose management strategy that simultaneously preserves biodiversity while allowing for some extractive resource use (Sumaila et al. 2000; Bohnsack et al. 2004). This practical and more compromising approach is now considered integral to sustainable fisheries management (Attwood et al. 1997; Denny et al. 2003).

Effects of fishing

Any form of exploitation can potentially cause cascading ecosystem effects even if managed (Barrett et al. 2007; Götz et al. 2009). The direct removal of target species for example, can alter the regulation of assemblage structure in the trophic web, which can impact overall community structure (Götz et al. 2009). Understanding the processes that structure these communities is essential to evaluate how they may respond to fishing.

There has been much debate over which mechanisms are important in different marine systems, as community structure varies in time and space, and is influenced by physical and biological factors (Pinnegar et al. 2000). On the one hand, it has been suggested that removing top predators, which can represent a large proportion of the total fish biomass in some waters, can cause a reduction in productivity (Dulvy et al. 2004; Shin et al. 2005; Barrett et al. 2007; Hutchings et al. 2009). Conversely, reducing predation may increase the abundance of prey and can therefore lead to an increase in biomass (Willis & Anderson 2003; Götz et al. 2009). Where targeted stocks are omnivorous, their continued depletion may dramatically impact invertebrate and algal cover. This can result in reductions in gross and net primary productivity (Pinnegar et al. 2000; Götz et al. 2009). Furthermore, in selecting for larger sized fish, fishing can create a selection pressure that favours smaller fish, and in turn reduce the reproductive capacity of fish stocks (Attwood et al. 1997; Mosquera et al. 2000; Claudet et al. 2010). Other more subtle effects may include changes in life history traits: for example, reducing the age at which species mature or change sex (*e.g.* Buxton 1993), commanding alterations in behaviour such as selecting against aggressiveness, or altering migration patterns (Attwood et al. 1997). Consequently, sustained fishing pressure does not simply reduce the abundance of targeted species. It has the potential, whether regulated or not, to impact all of the oceans' fish populations (Attwood et al. 1997; Götz et al. 2009).

Although MPAs have been documented to alleviate these pressures (Sumaila et al. 2000; Barrett et al. 2007), it is the no-take zones, areas that prohibit all forms of extractive activities, which offer the greatest protection to marine ecosystems (Lester et al. 2009).

No-take zones

By removing fishing pressure, no-take zones serve as natural refugia to all marine life within their boundaries (Attwood & Bennett 1995; Bohnsack et al. 2004). As such, they not only accelerate the recovery of depleted target species, but also eliminate the mortality of non-

target species or by-catch species. They prevent the degradation of habitats caused by destructive fishing practices and increase the probability of communities to persist (Bohnsack 1998; Murray et al. 1999; Götz et al. 2009; Currie et al. 2012). No-take zones also promote adult and larval 'spill-over' (net emigration across boundaries), which can enhance adjacent fisheries and act as a buffer against environmental stochasticity (Attwood et al. 1997; Guénette et al. 1998; Bohnsack et al. 2004). Furthermore, no-take zones facilitate fisheries management, serving as a valuable reference against which areas open to fishing can be assessed (Bohnsack et al. 2004). No-take zones can also provide baseline information on community assemblages, population parameters, and catch rates, as well as an understanding of how natural drivers, such as climate change, can affect marine resources (Griffiths 2000; Götz et al. 2011). In distinguishing between the effects of natural variability versus anthropogenic impacts, no-take zones can enhance scientific knowledge of the natural structure, function and performance of ecosystems. Understanding this is paramount for effective marine resource management (Bohnsack 1998; Hilborn et al. 2004).

Despite their inherent advantages, MPAs only protect 2.3% of the oceans (Spalding et al. 2012) and of these, only 0.08% are no-take zones (as the most recent data suggests) (Wood et al. 2008). South Africa, however, having designated over 20% of its coastline as MPAs and 11% of these as no-take zones (WWF 2011), is committed to implementing effective marine ecosystem management (Branch & Clark 2006).

South Africa's inshore marine resources

The transition along South Africa's coastline, from the cool temperate waters of the Atlantic Ocean to the subtropical Indian Ocean, has resulted in diverse marine biodiversity that has long supported productive marine fisheries (Gell & Roberts 2003; WWF 2011). However, increasing pressure from growing coastal communities, the introduction of motorised vessels, and inefficient conventional regulations, have resulted in overfishing and the collapse of 70% of South Africa's commercial linefish stocks (Attwood & Bennett 1995; Griffiths 2000; WWF 2011; Currie et al. 2012). Assessments of inshore fish stocks have also been hindered by the absence of reliable historical data and uncertainty surrounding the post-release survival rates of by-catch (Gell & Roberts 2003; WWF 2011). Faced with such widespread collapses, South Africa has expanded its use of no-take zones to better manage and protect fish stocks.
The persistent targeting of South Africa's temperate reef fishes by all linefishing sectors has been of particular concern to conservationists and fisheries' managers (Griffiths 2000; Götz

et al. 2008). Reef fishes are an important natural resource (Currie et al. 2012), accounting for approximately 25% of South Africa's commercially-important fish stocks (WWF 2011). Owing to their slow growth, late maturity and longevity (>15 years), reef fishes are especially vulnerable to exploitation, and past studies have indicated dramatic declines in mean size, abundance and diversity (Buxton 1993; Sauer et al. 1997; Gell & Roberts 2003). Furthermore, attempts to rebuild reef fish stocks by enforcing bag and size limits have been ineffective. This is due to the fact that reef fishes exhibit high levels of residency and that many species are prone to barotrauma, a condition that can prove fatal even after fish are released back into the ocean (Buxton 1993; Griffiths 2000; Kerwath 2005; Götz et al. 2007).

South Africa's no-take zones

No-take zones in South Africa are a more cost-effective management strategy than conventional regulations (*i.e.* bag limits, quotas and size limits). This is because no-take zones simplify enforcement as the no-take provision applies to an entire demarcated area, making infringements easier to detect (Attwood & Bennett 1995; Bohnsack et al. 2004). In addition, linefishermen prefer the clear set rules that no-take zones apply (Sauer et al. 1997; Bohnsack et al. 2004). As such, no-take zones require less manpower to enforce compliance than conventional regulations (Attwood & Bennett 1995).

South Africa has protected a substantial part of its marine biodiversity in a series of 35 marine reserves situated along its coastline (DEAT 2008). Despite many of the reserves being delineated in an *ad hoc* and opportunistic manner, and all varying in size and design, their placement across several different biogeographical regions affords considerable protection, creating a spatial network that functions as a whole (Hockey & Branch 1994; Turpie et al. 2000).

The most common effect of protection on fish in South Africa has been an increase in abundance and mean size (Attwood et al. 1997). For many of South Africa's endemic reef-dwelling sparids (seabreams), comparative studies have revealed that they are more abundant, later maturing and attain a greater maximum size inside than outside no-take zones (Buxton & Smale 1989). Sampling of subtidal reefs in the Tsitsikamma National Park (TNP), one of South Africa's largest (350 km2) and oldest no-take zones (49 years), revealed that roman (*Chrysoblephus laticeps*) and red steenbras (*Petrus rupestris*) respectively were 4 × and 13 × more abundant inside (Buxton & Smale 1989). In a similar study by Cowley et al. (2002),

comparisons of intertidal catch-per-unit-effort (CPUE) data obtained from TNP, found the abundance of four species, three sparids and one dischistid, to be 5-21 times greater inside the no-take zone than outside. Furthermore, in both studies concerning TNP, size frequency distributions showed a higher proportion of larger fish within the no-take zone and smaller maximum sizes in surrounding open-access areas (Buxton & Smale 1989; Buxton 1993; Cowley et al. 2002; Gell & Roberts 2003). These results are similar to those in other studies conducted around South Africa's tropical and temperate MPAs (Bennett & Attwood 1991; Bennett & Attwood 1993; Götz et al. 2009; Currie et al. 2012).

As fecundity typically increases exponentially with fish length, the higher proportion of larger size classes observed within no-take zones also directly improves the reproductive capacity of populations (Buxton 1993; Attwood et al. 1997). By allowing fish to reach their peak fecundity, egg production can be maximised (Attwood et al. 1997). This accelerates stock recovery within the no-take zone and in surrounding fishing grounds, through self-seeding and the export of eggs and larvae (Attwood et al. 1997). Based on the surface currents measured in the TNP, it was estimated that larvae dispersed as far as 580 km in only 30 days (Attwood et al. 2002). More recent studies suggest that larval dispersal may account for the high levels of genetic mixing found in populations of roman along South Africa's coastline despite their philopatric nature (Teske et al. 2010). The protection afforded to many reef fish within no-take zones presents a strong case for their use as an insurance against recruitment failure (Buxton 1993). Protection of the larger size classes within fish populations is particularly important for species that exhibit sequential hermaphroditism. Removing the larger fish can disrupt the gender balance of a population and thus decrease reproductive output (Buxton 1993). A study by Buxton (1993) showed that no-take zones play a significant role in maintaining a healthy sex ratio for dageraad (*Chyrsoblephus cristiceps*); a 3.8:1 female to male ratio within the TNP compared to 24:1 at some open access sites between Knysna and Mossel Bay. In preserving a demographic environment in which the plasticity of life history traits of fish can develop naturally, no-take zones enhance the survivorship and reproductive output of some of South Africa's most vulnerable reef fishes, many of which are the primary targets of the linefishing sector (Buxton 1993; Gell & Roberts 2003).

Due to their ecosystem objective, no-take zones also support greater fish diversity, and differing patterns of community composition, when compared to areas under fishing pressure (Prochazka 1998; Götz et al. 2009; WWF 2011). These findings have been confirmed in the

Goukamma MPA (Götz et al. 2009) and the TNP (Bennett et al. 2009) using underwater visual census (UVC). However, the relationship between diversity, community structure and protection is complex and difficult to predict (Palumbi 2001). The intermediate disturbance hypothesis (IDH) suggests that this relationship is likely to be dependent upon the level of human disturbance outside a no-take zone, and the level of predation in the ecosystem (Lester et al. 2009). The IDH states that when a disturbance, in this case fishing, is neither too rare nor too frequent, species diversity can be maximised, as competitively dominant and rapid colonising species can coexist (Svensson et al. 2012). At both extremes of the disturbance range, diversity remains low, owing to competitive exclusion and localised extinction (Svensson et al. 2012). Studies by Sandin et al. (2008) that examined reef fish density and diversity over a gradient of human disturbance corroborate this theory. Under minimal disturbance, the density and diversity of fish increases until top predators attain a threshold after which their prey declines. As disturbance continues to increase, diversity declines further, and the fish community structure shifts from one dominated by top-level predators to one characterised by lower trophic consumers. It is therefore evident that if no-take zones are to restore the integrity and resilience of ecosystems, they must not only augment fish numbers, but also maintain the sensitive ecological interactions that promote a naturally functioning community.

No-take zones can differ in their response to protection depending on their location, size and number (Sumaila et al. 2000). In particular, the value of one single large versus several small sized (SLOSS) no-take zones is often debated (Baker 2000). By preserving a greater and contiguous area (entity) rather than several smaller sized zones, large no-take zones are considered more likely to promote greater genetic diversity because they contain larger numbers and a variety of species (Baker 2000). For those species that are heavily fished, large no-take zones may also help maintain genetic variation by preventing numbers from becoming 'critically low' (Baker 2000). Furthermore, large no-take zones may be more advantageous to those species that require larger habitat areas in which to complete all stages of their life cycle (Attwood & Bennett 1995). However, the choice over the size of a no-take zone is rarely based between one single large and several small, but rather on which is more feasible (Attwood & Bennett 1995). Despite the advantages of large no-take zones, studies have shown that even those that are small in size can be beneficial (Attwood & Bennett 1995). For example, several small no-take zones, less than 10 km in size, are considered optimal for the protection of blacktails (*Diplodus sargus capensis*), a species found to

disperse no more than 30 km based on tagging data (Attwood & Bennett 1995). For nomadic species like galjoen (*Dichistius capensis*), even protecting a portion of these populations across their range in several small no-take zones may prevent their genetic variation from being compromised in areas where fishing pressure is high (Attwood & Bennett 1995). Furthermore, according to Roberts & Polunin (1991), a network of small no-take zones may be more effective for fish populations scattered over numerous reefs, as the greater 'perimeter-to-area' ratio may maximise adult 'spill-over' into adjacent fisheries. With respect to yield, no-take zones will only become counter-productive if they occupy an area larger than the total fishing area (Attwood & Bennett 1995).

Problems with reef fish monitoring and assessments

Despite positive biological responses of reef fish to protection (Bennett & Attwood 1991; Attwood & Bennett 1994), results should be interpreted with caution. Assessments on biological resources inside no-take zones are often focused solely on the status of target species (Barrett et al. 2007), with few studies evaluating the status of non-target species, especially those that are cryptic or nocturnal (*e.g.* Burger 1990; Prochazka 1998). Although these economically important species are often the *raison d'être* of reserves, the state of biodiversity as a whole should be considered when assessing their efficacy (Barrett et al. 2007).

Assessments between study sites or time periods often lack comparability, limiting the conclusions that can be drawn (Willis et al. 2003). A feature common to no-take studies is that insufficient data exists on fish populations before implementation, thereby excluding powerful 'before and after comparisons' (Willis et al. 2003; Edgar et al. 2004). This not only makes it difficult to document the effectiveness of a no-take zone over time, but is particularly concerning when no-take zones are supposed to function as a control against which exploited areas can be compared (Willis et al. 2000). In the absence of such data, management decisions may be based on misleading information (Griffiths 2000; Willis et al. 2003).

Predictions of fish stock recovery within no-take zones may also be confounded by the displacement of fishing effort to surrounding areas (Molloy et al. 2009). By reducing the available area that can be fished but not the number of boats, fishermen are forced to compete for fish in a smaller available area often further away from access points, whilst still trying to

retain the same yield and efficiency as before (Molloy et al. 2009). Many fishermen are also known to 'fish the line' of the no-take boundary, in the hopes that adult fish will 'spill-over' from the reserve into surrounding exploitable areas (Pichegru et al. 2011). These shifts in fishing patterns can cause stocks to decline more rapidly outside of no-take zones leading to a false sense of the protection provided by no-take zones (Molloy et al. 2009; WWF 2011). Without a measure of fishing intensity and how it changes over time and space, any differences revealed between a no-take zone and an exploited area cannot strictly be attributed to protection (Willis et al. 2000; Molloy et al. 2009).

Many studies also lack comparability because no-take zones and respective exploited areas were different in important characteristics such as depth, water temperature and topographic complexity (Willis et al. 2003). Finding any two sites that are similar in terms of their habitat, area and history is especially difficult when study areas are located long distances apart (Willis et al. 2003). However, depth, water temperature and reef profile are considered to be the most important predictors of reef fish distribution and abundance, and should therefore be controlled for when possible (Buxton & Smale 1989; De Vos 2012). Reef depth and profile determine the availability of food as well as provide refugia from predators and competitors, whilst temperature is suggested to influence fish mobility and physiology (Buxton & Smale 1989; Friedlander & Parrish 1998). Such variation in the spatial distribution of fish could therefore hinder the ability to detect changes in density between sites (Willis et al. 2000). Monitoring programs should therefore collect data across appropriate habitat types to account for environmental variation (Götz et al. 2009). In addition, understanding the association between reef fish and the habitats they occupy may benefit future reserve design (Friedlander & Parrish 1998; Huntington et al. 2010).

Numerous sources of natural variability further complicate attempts to measure change in the abundance, mean size, or diversity of fish communities (García-Charton 2000; Bennett 2007). Variation, stemming from annual changes in recruitment, migration patterns, and mortality, may cause temporal and localised alterations in the abundance and diversity of fish (Bennett 2007). Unless this variation is monitored over the long term, it may confound results (Ault & Johnson 1998; Bennett 2007). To account for these biases, it is necessary to identify the natural levels of variation so that any real changes can be detected (Bennett et al. 2009).

Sampling-associated variability is another important source of bias. Comparative studies between no-take zones and exploited areas may produce inaccurate results depending on reef

fish behaviour, and the skills and experience of the surveyor (Barrett et al. 2007). Many studies that employ CPUE data also fail to reference the number of anglers used or the duration of time spent at each fishing station (Willis et al. 2000). With no information on absolute effort, results from these studies are limited in their comparability (Bennett 2007). Furthermore, the extractive nature of these techniques, as well as those that use ichthyocides and explosives, raises concerns about their applicability in no-take zones (Willis et al. 2000).

Underwater visual censuses that use divers are often employed to enumerate the abundance and composition of reef fish in a non-destructive way (Willis et al. 2000; Watson et al. 2010). The method is advantageous because of its ability to document a high number of pelagic and benthic species, as divers are able to search all possible refuges in a designated area (Watson et al. 2005; Stobart et al. 2007; Bassett & Montgomery 2011). However, numerous sources of error can be associated with this sampling type including the misidentification of species, inaccurate counts of individuals, especially when larger shoals are present, inter-observer variability resulting from the speed at which a diver completes a survey and the diver's experience (Watson et al. 2005; Langlois et al. 2010). SCUBA diving surveys may also attract species that are bold, or deter those that are shy from being counted (Bennett 2007).

In an attempt to overcome these limitations, a permanent visual record of fish observed during surveys can now be achieved using video camera systems (Langlois et al. 2010; Watson et al. 2010). However, diver operated videos (DOVs) remain constrained by depth and the amount of time a diver can spend underwater (Bennett 2007; Stobart et al. 2007). In South Africa's high-energy marine environment, adverse environmental conditions and low visibility also often limit the number of days that are suitable for diving, reducing the number of samples that can be achieved (Bennett 2007; Pelletier et al. 2012). Furthermore, the high costs and logistical demands of UVC reduce the potential for their inclusion in sustainable long term monitoring programs (Bennett 2007).

Baited remote underwater video assessments

Baited remote underwater video (BRUV) surveys offer an alternative and complementary technique to UVC. The concept of the BRUV system is that reef fish are attracted into the field of view of an underwater camera using bait (Cappo et al. 2006). The video footage is then brought ashore and analysed for information on species composition, relative species abundance and diversity, all measures which are fundamental to monitoring protected areas

(Willis et al. 2000). This method minimises observer-biases, collects data with lower variance and maximises sampling efficiency, as multiple camera systems can be in use within the same deployment (Cappo et al. 2004; Cappo et al. 2006; Langlois et al. 2010; De Vos 2012). Methodological comparisons have shown that BRUVs, on average, record more reef species, higher species richness, and a greater number of mobile predators than UVC (Watson et al. 2005; Cappo et al. 2006; Harvey et al. 2007; Colton & Swearer 2010). Importantly, with the ability to achieve less variation in replicate samples within habitats, BRUVs increase the statistical power of tests to detect spatial and temporal changes in fish assemblages, thereby reducing the necessary sampling effort (Cappo et al. 2004; Harvey et al. 2007; Bernard & Götz 2012).

Several studies have suggested that BRUVs remain disadvantaged by their reliance on water clarity, and safe environmental conditions for deployment (Cappo et al. 2006; Colton & Swearer 2010). Yet, BRUVs can provide data in less than one metre visibility (L. De Vos, Save Our Seas, *pers. comm.*) and can sample in considerably poorer sea conditions than UVC (Cappo et al. 2006). Furthermore, unlike other methods, BRUVs are not restricted by depth and are neither extractive nor do they cause major disturbance to substrata and its epibenthos (Cappo et al. 2006). In addition, BRUV surveys require less manpower than other survey methods and are significantly safer to carry out than any diving operation (Willis et al. 2000). The time required to analyse video footage is also regarded to be excessive (Willis et al. 2000; Stobart et al. 2007). However, permanent video records remove the need for experts to conduct the fieldwork and allow impartial and repeatable measurements to be made, thereby standardising data collection and analysis (Cappo 2010). The use of bait presents another potential problem because there is no effective means to estimate the area from which the fish are attracted (Watson et al. 2005). For this reason, relative abundance can only be given rather than an absolute measure of fish density. Despite this shortfall, these measurements are in fact similar to CPUE measures (Willis et al. 2000; Watson et al. 2010).

Although BRUV technology is a popular monitoring tool in marine research, it was only first deployed in South Africa in 2008 in TNP (Bernard & Götz 2012) and in False Bay in 2011 (L. De Vos, Save Our Seas, *pers. comm.*). These studies determined that BRUV technology is a scientifically sound monitoring tool for documenting temperate reef fish assemblages at depths of up to 50 m. Since its first deployment in South Africa, BRUV technology has evolved rapidly and cameras are now more affordable and offer a higher picture resolution. Following suggestions made by De Vos (2012), a new BRUV system is now available

comprising a GoPro® HERO 2 HD camera (Woodman Labs 2009) in a dive housing, mounted horizontally at one end of a steel rig. On the other end, at a distance of one metre, is a bait canister, made from PVC plumbing pipe. The rig, attached to a buoy by means of a floating rope, is deployed from a small powered boat and left to film on the seafloor for a period of one hour. Multiple BRUV rigs can be deployed simultaneously to maximise sampling efficiency.

Long-term monitoring technique

Typically, monitoring protocols recommend the use of multiple techniques to accurately assess reef fish communities (Bennett et al. 2009; Götz et al. 2009). However, the financial and logistical limitations of such combined approaches make it unfeasible for responsible agencies to conduct monitoring programs in no-take zones over the long term (Caughlan & Oakley 2001; Stobart et al. 2007). These financial constraints together with the complexity and multi-species nature of South Africa's inshore fisheries make monitoring a daunting task. As a result, less than 3.5% of commercially-important species in South Africa are adequately assessed with respect to their conservation status (C.G. Attwood, UCT, *pers. comm.*).

Concern for the sustainability of South Africa's inshore fishing industries continues to grow. It is essential therefore that the no-take zone network be monitored regularly to assess its ability to achieve conservation and fisheries' management goals. For this purpose, BRUVs offer a standardised and repeatable technique with which to monitor reef fish that is more cost-effective, and easier to implement, than UVC or controlled angling surveys (Bernard & Götz 2012). Furthermore, BRUV footage is also a valuable visual aid that can be used to encourage support for marine conservation (De Vos 2012). Being non-destructive and non-extractive, BRUVs are ideally suited to monitoring within no-take zones (Bernard & Götz 2012).

CHAPTER 2

Patterns in reef fish assemblages as determined by baited remote underwater video (BRUV) along the western side of False Bay: effects of site, depth and protection

INTRODUCTION

Unsustainable exploitation has reduced many of South Africa's commercially-important linefish populations to less than 10% of their historic levels (Griffiths 2000). The persistent targeting of reef fish species is of particular concern to fisheries' managers and conservationists as their slow growth, late maturity, and philopatric nature makes them especially vulnerable to overexploitation (Buxton 1993; Griffiths 2000; Götz et al. 2008). Many once abundant species such as dusky kob (*Argyosomus japonicus*), galjoen (*Dichistius capensis*), geelbek (*Atractoscion aequidens*), roman (*Chrysoblephus laticeps*), red stumpnose (*Chrysoblephus gibbiceps*), and white steenbras (*Lithognathus lithognathus*) are considered collapsed, whilst red steenbras (*Petrus rupestris*) and seventy-four (*Polysteganus undulosus*) are deemed commercially extinct (C.G. Attwood, UCT, *pers. comm.*; Griffiths 2000; Yemane et al. 2004).

In response to evidence of overfishing, a conservation emergency was declared in South Africa's linefish industry in 2000 (MLRA, Section 16). This emergency measure placed a number of catch-and-effort restrictions and increased the number of marine protected areas (MPAs) and no-take zones (Branch & Clark 2006; Sowman et al. 2011).

The concept of protecting areas from exploitation is not new (Attwood et al. 1997). Areas too deep or dangerous for fishing once received natural protection from exploitation and likely served as critical refuges that supported surrounding fisheries through larval and adult spill-over (Bohnsack 1998; Gell & Roberts 2003). Recent advances in technology have equipped fishermen with better detection capabilities, and the ability to travel further to obtain their catch (Bohnsack 1998). This together with an ever-increasing pressure on marine resources has meant that these natural refuges have all but disappeared (Bohnsack 1998). By formally protecting ecosystems in their entirety, no-take zones are widely considered to be the

principal means by which marine resources can be sustainably harvested, and marine biodiversity conserved for future generations (Worm et al. 2009).

In spite of a long history of fishing, notable declines in populations of reef fish in False Bay have been observed in the last two decades (Attwood & Bennett 1995; Sauer et al. 1997). Conventional management restrictions such as bag and size limits have been difficult to enforce as multiple fishing sectors including recreational, subsistence and commercial fisheries, all target reef fish (Sauer et al. 1997; Griffiths 2000). In an effort to resolve this situation, the Table Mountain National Park (TMNP) Marine Protected Area (MPA) was proclaimed in 2004 in order to offer various protection levels including conventional management restrictions such as bag and size limits, closed seasons, and no-take zones (Hauck & Kroese 2006; Tunley 2009). Properly managed, the no-take zones have the potential to maintain the economic viability of fisheries, and their productivity, by conserving natural ecosystem function (Hilborn et al. 2004).

Effective management of no-take zones requires continuous monitoring of the fish communities within and around them, to evaluate the response of marine ecosystems to anthropogenic and environmental change (Pomeroy et al. 2005; Bennett et al. 2009). Since its proclamation however, no comprehensive monitoring programme has existed, making it difficult to evaluate if the TMNP MPA is achieving its biodiversity conservation and fishery management goals (Clark 2001). Comprehensive assessments have been hindered by the need to monitor over much broader spatial and temporal scales than funding agencies could afford (Carr 2000). Considerable challenges lie in the ability to monitor multiple species, each with different life-history characteristics, each targeted by different fisheries, using one single monitoring method (Sauer et al. 1997; Caughlan & Oakley 2001; Stobart et al. 2007). Furthermore, the small size of the no-take zones, in comparison with others in South Africa, has been a contentious issue among scientists who debate whether a single large or several small (SLOSS) no-take zones provide the optimal configuration to conserve biodiversity and restore fish stocks to sustainable levels (Attwood & Bennett 1995). With no baseline data, the human effects of exploitation on reef fish assemblages cannot be compared.

False Bay forms the western edge of the range for many temperate reef fish species (Smith & Heemstra 2003). Compared with populations at the core of their range, peripheral populations commonly exist at lower abundances. As such, these populations are of greater conservation concern as they are more vulnerable to external pressures such as exploitation, pollution, and

climate change (Vucetich & Waite 2003). Furthermore, due to their slow growth, the recovery of reef fish following the implementation of protective measures is often delayed (Babcock et al. 2010). Ensuring the protection of reef fish is important not only in terms of the food, recreation and tourism value they provide to coastal populations, but also to maintain their important ecological role (Götz et al. 2009; Currie et al. 2012).

The aim of this study was to use baited remote underwater video (BRUV) to describe patterns of diversity and abundance of reef fish along the western side of False Bay relative to site, depth, and protection. This study will also serve to provide baseline data on reef fish communities within the TMNP MPA. With South Africa's larger no-take zones already demonstrating positive conservation benefits to reef fish (Buxton & Smale 1989; Bennett & Attwood 1991; Götz et al. 2008), it is important to assess the protective value of smaller no-take zones.

MATERIALS AND METHODS

Study Area

Table Mountain National Park MPA is situated in False Bay in the Western Cape Province of South Africa (Figure 1). It is managed by South African National Parks (SANParks) and the Department of Environmental Affairs (DEA) within Oceans and Coasts (Clark 2001). Designated in 2004, this MPA protects 953.2km² of inshore marine habitat. The MPA stretches around the Cape Peninsula, from Green Point in Cape Town on the Atlantic Ocean side, to Bailey's Cottage in Muizenberg on the Indian Ocean side in False Bay (Tunley 2009). Owing to its unique location in the transition zone of the warm Agulhas and cold Benguela currents, TMNP MPA supports a rich diversity of marine life, which has long been exploited by commercial and small-scale fisheries (Spargo 1991; Griffiths 2000).

While regulated fishing is allowed in the majority of the TMNP MPA, there are also six restricted no-take zones, which collectively represent 5.9% of its total area (Table 1) (Clark 2001). This study focused on the four no-take zones located on the western side of False Bay (on the eastern side of the Peninsula): St. James, Boulders, Castle Rocks and Paulsberg (Figure 1 a-d). These no-take zones contain some of the historically least disturbed and least fished areas (Clark 2001).

Reefs on the western shore of False Bay extend to depths of 20 m until they reach the sand interface (Spargo 1991; van Zyl 2011). Aside from the St. James no-take zone in the north, which is located on Malmesbury shale, the three zones further south are all underlain by granite (Spargo 1991).

Table 1: Protection afforded by each of the four no-take zones under study in the Table Mountain National Park Marine Protected Area. The high water mark defines the boundary on land for each no-take zone. These data were drawn from Clark (2001), Kerwath (2005) and C.G Attwood (UCT, *pers. comm.*).

No-take zone	Coastline protected (km)	Area (km^2)	Year of promulgation
St. James	1.3	0.30	1979
Boulders	16.9	4.00	2004
Castle Rocks	3.7	3.00	1979
Paulsberg	3.8	3.00	2004

Experimental Design

Sampling Sites

Baited remote underwater video surveys were conducted in four sites along the western side of False Bay. For each of the four sites, two samples inside a no-take zone and two samples outside, in an adjacent exploited area, were taken.

The term 'site' refers to each location *i.e.* St. James, Boulders, Castle Rocks, and Paulsberg. For the duration of the study period, it was intended that 16 samples be collected at each site. This resulted in the collection of eight samples inside a no-take zone and eight samples outside in a respective exploited area. When defining samples by protection level, they will be referred to as no-take zone samples and exploited area samples.

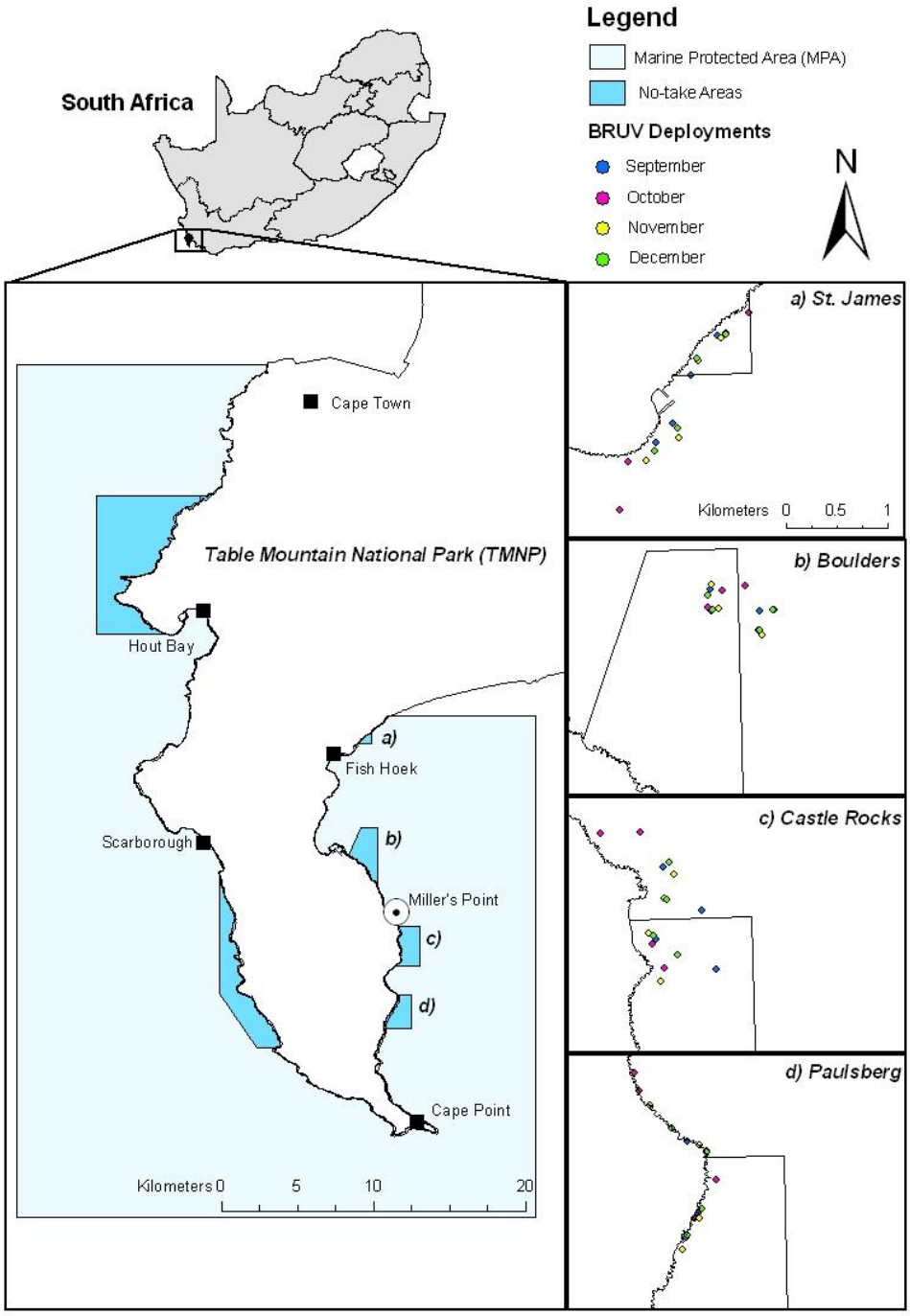

Figure 1: The Table Mountain National Park Marine Protected Area (light blue) and six no-take zones (dark blue). The four no-take zones to the east of the Cape Peninsula are: a) St. James b) Boulders c) Castle Rocks d) Paulsberg. Samples are indicated by colour per month (September (blue), October (pink), November (yellow), and December (green).

BRUV system

The BRUV systems comprised a weighted, mild steel rig on which a Go-Pro® HD Hero 2 camera (Woodman Labs 2009) was mounted to one end using a GoPro® bicycle clamp. At the other end, at a distance of one metre from the lens, a perforated PVC bait canister (130 mm x 110 mm with 10 mm perforations), secured using cable ties, lay 40 cm above the ground in the camera's field of view (FOV) (Figure 2). One kilogram of pilchards (*Sardinops sagax*), freshly crushed so as to maximise the dispersal of fish oil and flesh, was used as bait for each BRUV deployment (Langlois et al. 2010; Watson et al. 2010). Each rig was deployed on the seafloor (Figure 3), connected to the surface by means of floating rope (60 m) attached to a subsurface (10 cm in length x 3 cm diameter) and a surface buoy (30 cm in length x 10 cm diameter). To prevent entanglement with the camera system, the subsurface buoy was attached to the rope 1 m above the chain-weight. The BRUV systems were left to film on the seafloor for a period of one hour and then retrieved. Previous research has shown that one-hour deployments are advisable to record 95% of species diversity (Watson et al. 2010; Colton & Swearer 2010; Bernard & Götz 2012; De Vos 2012). The BRUV systems were given a minimum of five minutes settlement after which one hour of sampling time began.

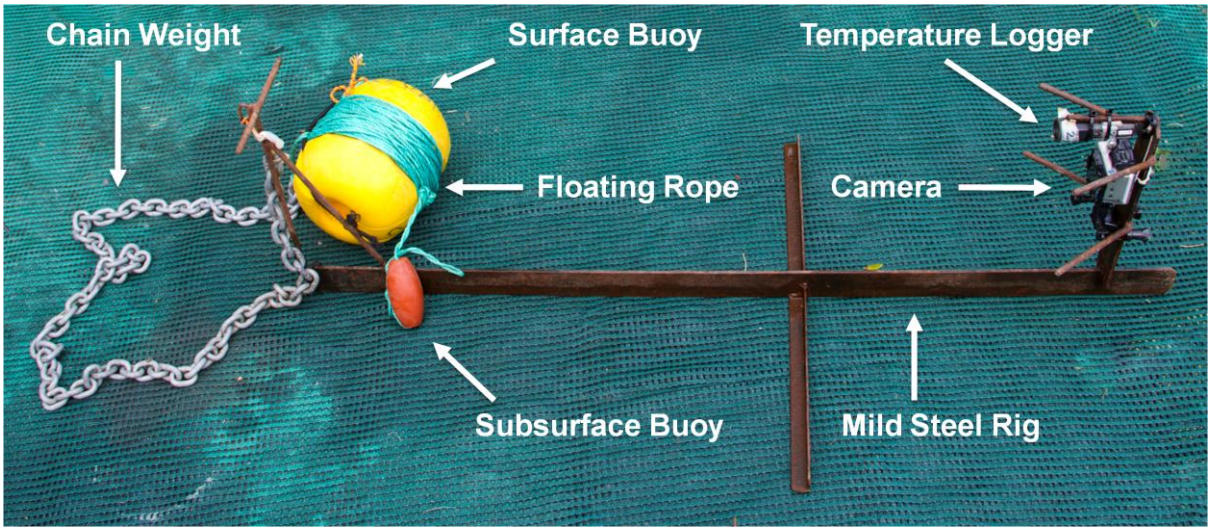

Figure 2: The baited remote underwater video system with all components except bait canister.

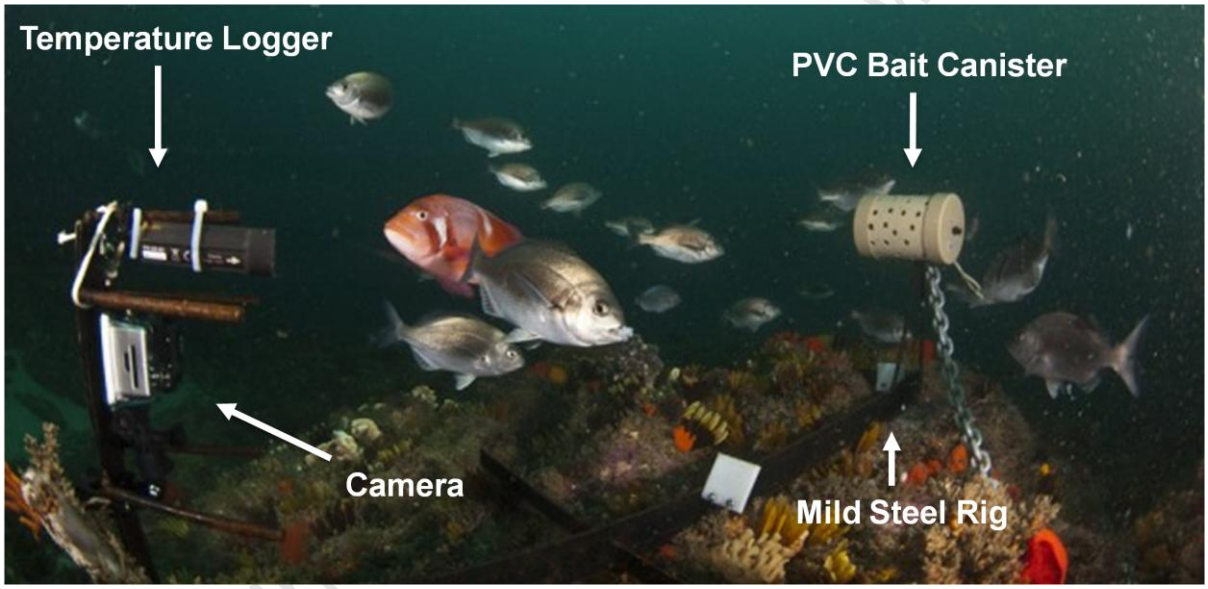

Figure 3: The deployed baited remote underwater video system with bait canister attached.

BRUV deployments

Four identical BRUV systems were employed to increase sampling efficiency. All four sites (St. James, Boulders, Castle Rocks, and Paulsberg), each with four samples, were completed over the course of the same day, once a month from September to December 2012. The four BRUV systems were deployed within 15 minutes of each other and filmed simultaneously for one hour. After each deployment, a new memory card and battery was installed and the bait canister replenished. The position of a sample was chosen randomly with the proviso that it was 200 m away from any other sample. A distance of 200 m was chosen to reduce the

likelihood of overlapping bait plumes and avoid pseudo-replication caused by fish moving between BRUV systems (Watson et al. 2010). Because bathymetric charts were unreliable as far as predicting the spatial extent of reef, the exact position of a sample was chosen with reference to the boats echo sounder and recorded using a Garmin Global Positioning System (GPS) device. In successive months, samples were chosen without reference to the previous month's positions and in this way it was deemed that the locations were chosen randomly. All BRUV deployments were conducted off an 8 m rigid inflatable boat.

Environmental variables

Temperature loggers (onset® HOBO® Water Temp Pro v2), attached to the rigs, recorded water temperature on the seafloor every five minutes during the entire study period. The temperature at each sample was determined from the start and end times of each deployment, and a median value taken for its duration (De Vos 2012).

The boats echo sounder was used to determine the depths at which deployments were made. Depths for each sample within a site were made within the same depth category to ensure comparability.

Video analysis

In order to standardise the analysis procedure, all video footage was reviewed by one researcher using VLC Media Player 2.02. Every fish observed on the videos was identified to species level, using references images where necessary from Smith & Heemstra (2003) and Heemstra & Heemstra (2004). The maximum number of individuals for each species counted in any one frame for the duration of a video was recorded as MaxN (Cappo et al. 2007; Harvey et al. 2012). The use of MaxN is considered a conservative estimate of relative abundance and avoids the possibility of recounting individuals that return into the camera's FOV (Willis et al. 2003; Watson et al. 2005).

Habitat classification

Reef profile, determined from the BRUV footage, was given a score from zero to ten based on the difference between the lowest and highest reef structure in the screen. Profile was classified for each sample as low if a score below 5 on the scale was given, or high for a score of 5 or above. Bottom-sediment type was classified as either reef or sand.

Data Analysis

Analyses were conducted to examine spatial trends in reef fish communities between no-take zones and exploited areas. In addition, analyses were performed to identify whether site, depth or protection level were influencing any of the observed differences. Data were analysed at the level of individual sites and all fish were identified to species level. This approach required employing univariate and multivariate analysis techniques.

Month was not factored into univariate or multivariate analyses as it was expected that reef fish assemblages would change over the four month study period (Bennett & Attwood 1993). The focus of this study was not to monitor changes between the sampled months but rather to determine the differences in reef fish composition and abundance between sites, depths and protection level (Westera et al. 2003).

Univariate Analyses

Environmental factors

A two-factor analysis of variance (ANOVA) was used to test for differences in depth and reef profile between site and protection level. Tukey (Zar 1984) post-hoc testing determined how these environmental factors influenced the differences between sites and protection level. The depth range was split into four factor levels: A (< 4 m), B (4 - < 8 m), C (8 - < 16 m) and D (16-32 m), and reef profile was classified as either 'low' or 'high' (Chapter 2, methods). Box plots were drawn to visually compare depth and reef profile measurements between no-take zones and exploited areas in each site. The variation in temperature recordings for the entire study period was also illustrated. All univariate analyses were performed with the statistical software R (version i386 2.15.0) (R Development Core Team 2012).

Species diversity

The Shannon-Wiener diversity index (H'), which incorporates components of, species richness and evenness, was chosen to express the diversity of reef fish communities:

$$H' = - \sum p_i (\log p_i)$$

where p_i is the proportion of the total number of individuals arising from the *i*th species.

This index was calculated for each sample (64 in total). To compare diversity between categorical factors (site, depth, and protection level), ANOVA was chosen. Reef profile could not be statistically tested as there was an inequitable distribution across the full range of profiles.

Frequency of occurrence and relative abundance

Species frequency of occurrence was calculated by counting the number of samples in which a species was recorded and dividing it by the total number of samples (64). Species frequency of occurrence was also determined for each site (16 samples) and between no-take zones and exploited area samples (8 samples for each level of protection). The relative abundance of each species was calculated by taking the sum of the MaxN values determined from the samples and dividing it by the total number of samples (64). The relative abundance was also calculated for each site and between no-take zones and exploited area samples. Cumulative species abundance curves were plotted against log species rank for each site. This provided a visual assessment of the species diversity of each site.

Relative abundance of four important reef fish species

Generalised linear models (GLMs) were applied to examine the effects of site, protection level, depth, and reef profile on the abundance (MaxN) of hottentot, roman, pyjama catshark, and red steenbras. These four species were chosen as they represent populations which are currently depleted and according to the Southern African Sustainable Seafood Initiative (SASSI), are vulnerable to collapse (Appendix Table A2). Generalised linear models were applied, rather than ANOVA, as the models needed to include both categorical and continuous variables. Furthermore, sample sizes differed among the random factors to the extent that it may have compromised the requirements of an ANOVA. Models were first fitted to a Poisson distribution and then tested for over-dispersion using Pearsons residuals. If Pearsons residuals were lower than 1, a Poisson distribution was chosen. If Pearsons residuals were greater than 1, a negative binomial model was applied to account for over-dispersion. As many different interactions were possible, a model was chosen that included only the interaction between site and protection to focus the analyses on the effect of protection on sites. Because an interaction between temperature, depth, and reef profile was meaningful, it was tested in other statistical analyses. Different models were assessed using the Akaike information criterion (AIC; Akaike 1973). A species MaxN value recorded for every sample

was used as an independent sample unit. Depth and profile were used as continuous variables and a value of 1 was added to each profile score to enable log-transformation. The response code 'MaxN' was defined as a count variable with Poisson distribution for roman, pyjama catshark, and red steenbras; however, a negative binomial distribution was the most appropriate for hottentot. The log-link was applied in all models. To model MaxN, the following parameters were combined into a GLM:

$$\ln(\text{MaxN}) = \alpha + \beta_{site} + \beta_{protection\ level} + \beta \times \ln(\text{depth}) + \beta \times \ln(\text{profile} + 1) + \ln(\text{error})$$

where site and protection level are categorical and depth and profile are continuous variables. A pseudo R^2 was calculated for each model to understand the variation explained by the model. This was calculated using the following equation:

$$\text{Pseudo } R^2 = 1 - (\text{residual deviance/null deviance})$$

Multivariate analyses

Conventional multivariate inferential methods that assume statistical normality such as MANOVA, were unsuitable for the data as distributions of abundances per fish species were highly skewed and contained many zeros (Anderson 2001). Data were therefore analysed using non-parametric approaches. All analyses were conducted using Bray-Curtis similarity matrices on the untransformed data. Although abundance data are usually transformed, untransformed data are more sensitive to changes in the abundance of species (Cowie et al. 2000; Widdicombe et al. 2004). In addition, this study was interested in the abundance of shoaling species as these are also targeted by the fishing industry and transformation would have down-weighted the importance of such shoaling species. For this study, it was expected that only incremental differences in fish abundances would be found as no-take zones were small in size. Furthermore, as BRUV systems were programmed to film for one-hour at each deployment, abundance data was already standardised. The Max N measure also avoided double counting. All multivariate analyses were performed using PRIMER-E v6 (Clarke & Gorley 2006).

Permutational multivariate analysis of variance (PERMANOVA) was used to compare variation in reef fish assemblage structure for three interactions. These interactions included site x protection level, site x depth and depth x protection level. The factors 'site' and 'protection level' were regarded as fixed and depth as a random effect. The factor 'protection

level' combined samples from all sites within the no-take zones compared with those collected within exploited areas. Data were classified using two categorical variables 'protected' or 'unprotected.' PERMANOVA were carried out by using a pseudo-F statistic with 999 random permutations of the data. The aim of this analysis was to determine if reef fish assemblages were influenced more by site or depth and if protection level influenced the number of individuals recorded. PERMANOVA was performed using the extension software PERMANOVA+ in PRIMER-E v6 (Clarke & Gorley 2006).

To test for differences in species composition between the categorical factor levels of site, depth, and protection level one-way Analysis of Similarity (ANOSIM) tests were performed on each variable separately with 999 permutations (Clarke 1993). To find natural groupings of samples according to site, depth and protection level, cluster dendrograms using the group average method were drawn (Clarke & Warwick 2001). Multidimensional scaling (MDS) plots were created to visually display the differences determined with ANOSIM. When ANOSIM results detected significant differences between factors, ($p < 0.05$) a similarity percentage (SIMPER) was calculated to identify the contribution of an individual species towards these differences (Clarke 1993; Clarke & Warwick 2001). A cut-off criterion was applied to determine a subset of species that cumulatively contributed to 50% of the dissimilarity value.

RESULTS

Sample size and distribution

Sixteen BRUV deployments could be achieved in a single day every month. A total of 64 BRUV samples were collected across the four sites. All deployments were made during the day between 08:00 and 16:00. Poor weather conditions did not disrupt the sampling design but it did influence the choice of sampling days. Although the aim was to conduct sampling at the end of the month, as was achieved in September and October, the November and December samples were three weeks apart. All deployments landed on reef habitat, except for one in St. James which landed on sand. Monthly samples around Boulders, Castle Rocks and Paulsberg were conducted in closer vicinity to each other as reef areas were less extensive than around St. James (Figure 1).

Univariate Results

Environmental factors

Site depths ranged from 2.6 m to 22.0 m among sites (Appendix Table A1). Samples for each pair of no-take zone and respective exploited area were on average shallowest at St. James (5.88 m ± 2.14 SD) (Figure 4a), followed by Paulsberg (8.32 m ± 2.33 SD), Castle Rocks (11.05 m ± 2.04 SD) and Boulders (16.4 m ± 2.67 SD). Depth proved to be significantly different among sites but not between protection levels (Table 2), except for Paulsberg's no-take zone and its respective exploited area, which led to a significant interaction effect.

A total of 46 samples were classed as high and 18 as low profile reef (Appendix Table A1). There was a significant difference in reef profile between St. James and Paulsberg, but not among any other sites (Table 2). Paulsberg had the highest frequency of samples classified as high profile compared with St. James that had the lowest (Figure 4b). There was no significant difference in reef profile between no-take zones and their respective exploited areas.

Water temperature, measured on the seafloor during camera deployments, ranged from 15°C to 18°C during the study period (Figure 5). Water temperature was lowest in September/October and highest in November/December. Despite monthly variation among some sites, there was no significant difference in temperature among sites and between protection levels (Table 2).

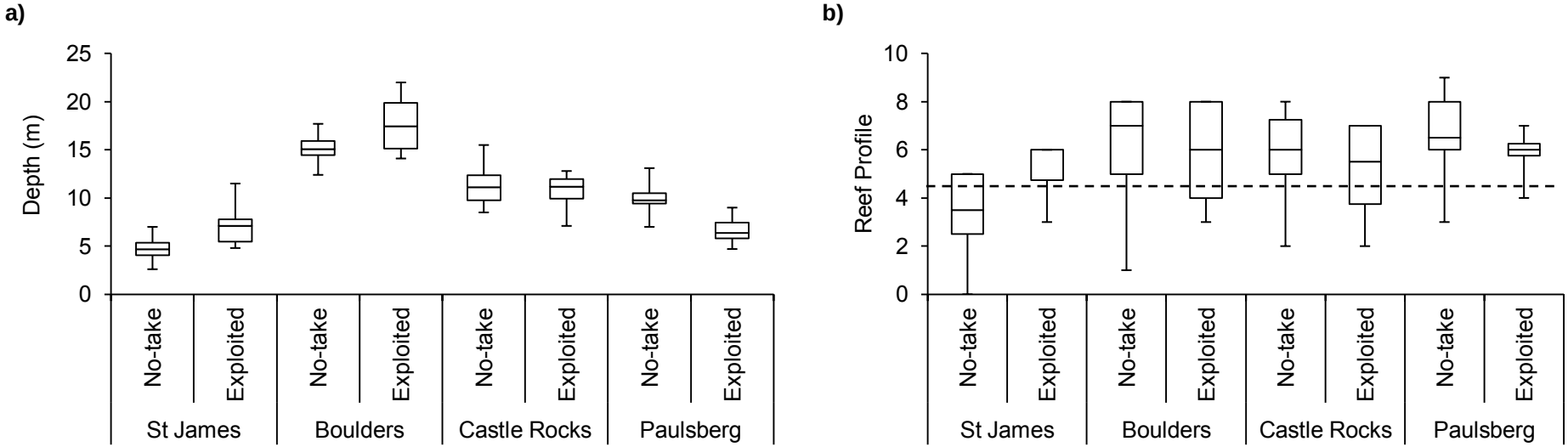

Figure 4: Depth (a) and reef profile (b) recorded in the no-take zones and exploited areas during baited remote underwater video surveys in the Table Mountain National Park Marine Protected Area. Reef profile is given a score out of ten. The broken line in (b) separates deployments made on either low (≤ 4.5) or high profile reef (≥ 5). The mean, first and second quartiles, and error bars representing the maximum and minimum values recorded are shown on both graphs.

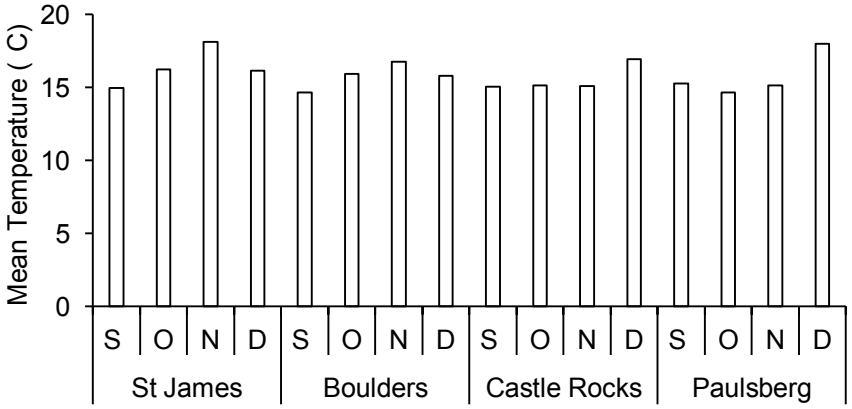

Figure 5: Mean water temperature measured across the four sites (no-take zones and exploited areas combined) over September (S), October (O), November (N) and December (D). Standard deviations bars are not included as there was only minimal variation in the temperature data.

Table 2: Summary of two-factor ANOVA of environmental variables testing the effects of site (St. James, Boulders, Castle Rocks and Paulsberg), protection level (no-take zone and exploited area), and their interactions.

Metric	Source of Variation	Df.	SS	F	p	
Temperature	Site	3	5.790	1.566	0.208	n.s.
	Protection level	1	0.040	0.033	0.856	n.s.
	Site x Protection level	3	0.202	0.164	0.920	n.s.
	Residuals	56				
Depth	Site	3	978.800	82.484	<0.001	***
	Protection level	1	1.000	0.240	0.626	n.s.
	Site x Protection level	3	97.600	8.223	<0.001	***
	Residuals	56	221.500			
Profile	Site	3	36.190	3.291	0.027	*
	Protection level	1	0.250	0.068	0.795	n.s.
	Site x Protection level	3	20.250	1.842	0.150	n.s.
	Residuals	56	205.250			

*** $p < 0.001$
* $p < 0.05$
n.s. = not significant

Species diversity

A total of 36 species from three marine classes and 18 families was recorded (Appendix Table A2). Twenty-six species formed part of the class Actinopterygii, nine species were from the class Chondrichthyes and one from the class Myxini. The most common family were sparids (seabreams) contributing 13 species in total. The rarest families were comprised of only one species.

Fourteen families were observed in Castle Rocks compared to 11 families in St. James, Boulders, and Paulsberg. Castle Rocks also had the highest total number of species recorded of all the sites (27 species), followed by Paulsberg (26 species), Boulders (24 species) and St. James (21 species). The same ranking was observed in respect of the Shannon-Wiener diversity index with higher species diversity in Castle Rocks and Paulsberg compared with St. James and Boulders (Figure 6). These differences were significant between St. James and Castle Rocks, St. James and Paulsberg, and Boulders and Paulsberg (Table 3).

There was no significant difference in the Shannon-Wiener diversity index between no-take zones and exploited areas (Table 3). The relationship between diversity and protection levels was not consistent (Figure 6). The Shannon-Wiener diversity index was higher in the

exploited areas of St. James and Boulders than inside their respective no-take zones (Figure 6). In Castle Rocks and Paulsberg, however, the diversity index was higher inside the no-take zones than in their respective exploited areas. No significant influence of depth on diversity was found (Table 3). Raw data for the Shannon-Wiener diversity index can be found in Appendix Table A3.

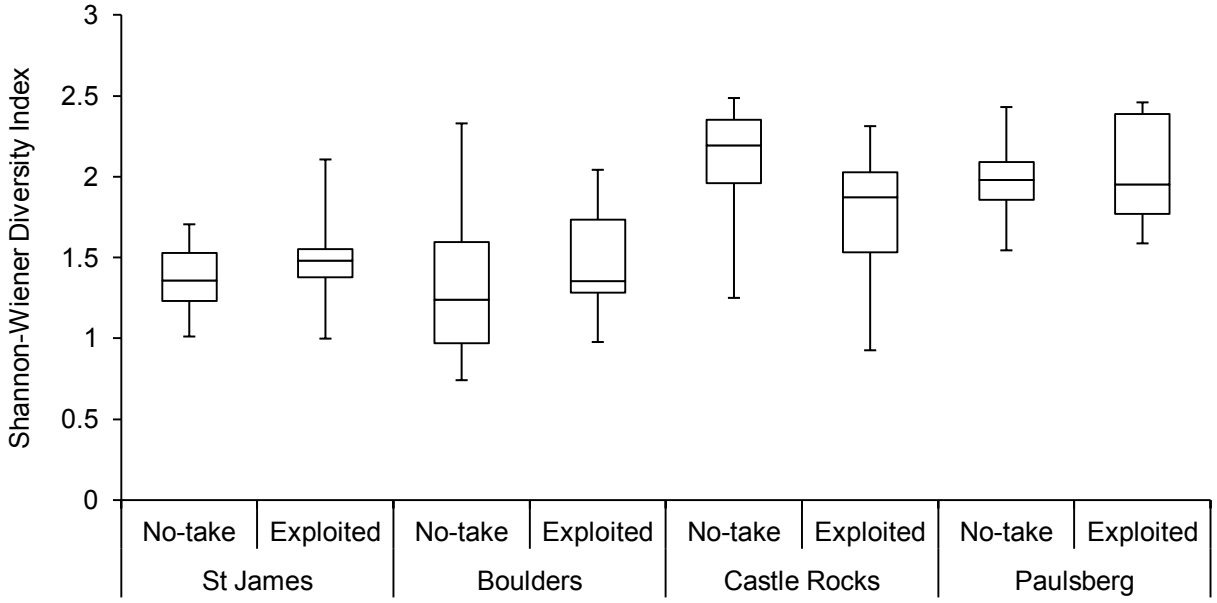

Figure 6: Box-plots comparing the Shannon-Wiener diversity index among the no-take zones and exploited areas of the Table Mountain National Park Marine Protected Area recorded in baited remote underwater video surveys over four months. The mean, first and second quartiles, and error bars representing the maximum and minimum values recorded.

Table 3: Summary of ANOVAs on Shannon-Wiener diversity indices testing the effects of site (St. James, Boulders, Castle Rocks and Paulsberg), protection level (no-take zone and exploited area), depth categories (A (< 4 m), B (4 - < 8 m), C (8 - < 16 m) and D (16-32 m)), and their interactions.

Metric	Source of Variation	Df.	SS	F	p	
SW Diversity	Site	3	12.922	9.389	< 0.001	***
	Protection level	1	0.058	0.126	0.724	n.s.
	Site x Protection level	3	1.296	0.942	0.427	n.s.
	Residuals	56	25.692			
	Site	3	12.922	9.022	< 0.001	***
	Depth	3	0.174	0.121	0.947	n.s.
	Site x Depth	2	0.614	0.644	0.529	n.s.
	Residuals	55	26.257			

*** p < 0.001
n.s. = not significant

Species occurrence and abundance

The most frequently observed family were sparids (40%), followed by scyliorhinids (27%), cheilodactylids (10%) and clinids (8%) (Table 4). The most frequently recorded species among sites was hottentot (*Pachymetopon blochii*) (91%), followed by roman (89%), and puffadder shyshark (*Haploblepharus edwardsii*) (83%). Five of 36 species including broadnose sevengill cow shark (*Notorynchus cepedianus*), dusky kob, eagleray (*Myliobatis aquila*), elf (*Pomatomus saltatrix*) and soupfin shark (*Galeorhinus galeus*), were only observed once.

St. James had the highest frequency of sea catfish (*Galeichthys feliceps*) (31%), strepie (*Sarpa salpa*) (25%), and white stumpnose (*Rhabdosargus globiceps*) (25%) of all the sites (Table 4). Blacktails (*Diplodus capensis*) (25%) and short-tail stingrays (*Dasyatis brevicaudata*) (13%) were seen the most frequently in St. James and Paulsberg. In comparison to the other three sites, puffadder shysharks were the least frequently observed in St. James (69%). Dusky kob (6%), elf (6%), and eagleray (6%) were only recorded in St. James.

Boulders had the highest frequency of occurrence of steentjie (*Spondyliosoma emarginatum*) (75%), leopard catshark (*Poroderma pantherinum*) (56%), Cape horse mackerel (*Trachurus*

capensis), (56%), jutjaw (*Parascorpis typus*), (38%), janbruin (*Gymnocrotaphus curvidens*) (38%), spotted gully-shark (*Triakis megalopterus*) (31%), redfingers (*Cheilodactylus fasciatus*) (31%), and barehead gobies (*Caffrogobius nudiceps*) (13%) of all the sites (Table 4). Boulders had the lowest frequency of dark shysharks (*Haploblepharus pictus*) (38%) among sites. In addition, there were no sightings of fransmadams (*Boopsoidea inornata*), six-gill hagfishes (*Eptatretus hexatrema*), blacktails, or short-tail stingrays in Boulders unlike the other three sites. There were also no sightings of red steenbras or red stumpnose in either Boulders or St. James. Boulders was the only site to yield pangas (*Pterogymnus laniarius*) (31%), bank steenbras (*Chirodactylus grandis*) (13%), and soupfin sharks (6%).

Castle Rocks had the highest frequency of roman (100%), fransmadam (75%), red stumpnose (25%), galjoen (19%) and zebras (*Diplodus cervinus hottentotus*) (19%) of all the sites (Table 4). Dark shysharks (94%) and super klipfish (*Clinus superciliosus*) (94%) were seen the most frequently in Castle Rocks and Paulsberg. In addition, evil-eye pufferfish (*Amblyrhynchotes honckenii*) were observed the most often in Castle Rocks and St. James (13%). Sea catfish (19%) were observed the least frequently in Castle Rocks. Castle Rocks was the only site that did not feature white stumpnose but was also the only site where broad-nose sevengill cowsharks (6%) were recorded.

Paulsberg had the highest frequency of hottentot (100%), pyjama catshark (*Poroderma africanum*) (88%), two-tone fingerfin (*Chirodactylus brachydactylus*) (75%), red steenbras (56%), six-gill hagfish (50%) and blue hottentot (*Pachymetopon aeneum*) (38%) of all the sites (Table 4). Cape knifejaws (*Oplegnathus conwayi*) were observed the most frequently in Paulsberg and Boulders (13%). Steentjie was observed the least frequently among sites in Paulsberg and St. James (13%). In addition, no Cape horse mackerel were recorded in either Paulsberg or St. James.

Eighteen species were observed more frequently in exploited areas than in no-take zones (Table 4). These formed part of seven families including sparids and scyliorhinids. The remaining 14 species that were observed more frequently in no-take zones formed part of ten families overall. The family carangid, of which Cape horse mackerel form a part, were observed with the same frequency in no-take zones and exploited areas. Castle Rocks was the only site to have a higher frequency of observations of fish inside its no-take zone (54%) than in its respective exploited area (46%). For Paulsberg (48% no-take zone vs 52% exploited area), Boulders (45% no take zone vs 55% exploited area) and St. James (40% no-take zone

vs 60% exploited area), fish were more frequently observed in their respective exploited areas.

Table 4: The relative abundance of species recorded using baited remote underwater video surveys (MaxN) in the no-take zones (NT) and exploited areas (EXP) of the Table Mountain National Park Marine Protected Area. The relative abundance (MaxN) of each species across 'All Sites' is provided together with the SD (standard deviation). 'Freq.' refers to the number of times each species was recorded. Species are ordered according to descending relative abundance across 'All Sites.'

			AVERAGE RELATIVE ABUNDANCE									
					ST. JAMES		*BOULDERS*		*CASTLE ROCKS*		*PAULSBERG*	
Family	Species	Freq.	***ALL SITES***	*SD*	*NT*	*EXP*	*NT*	*EXP*	*NT*	*EXP*	*NT*	*EXP*
Sparidae	Hottentot *(Pachymetopon blochii)*	58	**7.05**	9.17	2.75	2.38	8.25	19.13	5.38	5.13	8.75	4.63
Carangidae	Cape horse mackerel *(Trachurus capensis)*	6	**4.42**	15.47	0.00	0.00	23.63	8.00	0.00	3.75	0.00	0.00
Sparidae	Strepie *(Sarpa salpa)*	8	**4.06**	16.00	0.00	7.63	0.00	0.00	0.50	13.13	6.13	5.13
Sparidae	Steentjie *(Spondyliosoma emarginatum)*	27	**2.88**	8.00	0.00	4.13	2.38	13.25	1.13	1.88	0.13	0.13
Sparidae	Roman *(Chrysoblephus laticeps)*	57	**2.61**	1.78	1.88	2.38	1.25	3.75	4.63	2.75	1.75	2.50
Sparidae	Fransmadam *(Boopsoidea inornata)*	21	**2.53**	7.00	0.00	0.63	0.00	0.00	10.75	1.88	3.38	3.63
Scyliorhinidae	Puffadder shyshark *(Haploblepharus edwardsii)*	53	**1.91**	1.31	0.75	1.75	2.25	1.75	2.25	2.00	2.50	2.00
Scyliorhinidae	Pyjama catshark *(Poroderma africanum)*	41	**1.41**	1.52	0.13	0.50	1.88	1.50	2.25	1.50	1.38	2.13
Scyliorhinidae	Dark shyshark *(Haploblepharus pictus)*	43	**1.06**	0.96	0.50	0.75	0.50	0.38	1.75	1.25	1.38	2.00
Clinidae	Super klipfish *(Clinus superciliosus)*	50	**0.95**	0.68	0.38	1.38	0.75	0.63	1.25	1.13	1.13	1.00
Cheilodactylidae	Twotone fingerfin *(Chirodactylus brachydactylus)*	36	**0.75**	0.80	0.00	0.63	1.00	0.75	0.75	0.63	1.25	1.00
Scyliorhinidae	Leopard catshark *(Poroderma pantherinum)*	24	**0.50**	0.76	0.00	0.38	0.50	1.25	0.38	0.25	0.25	1.00

Table 4: *continued*

			RELATIVE ABUNDANCE									
			ALL		ST. JAMES		BOULDERS		CASTLE ROCKS		PAULSBERG	
Family	Species	Freq.	**SITES**	SD	NT	EXP	NT	EXP	NT	EXP	NT	EXP
Sparidae	Red steenbras *(Petrus rupestris)*	17	**0.44**	0.83	0.00	0.00	0.00	0.00	1.50	0.50	0.38	1.13
Cheilodactylidae	Redfingers *(Cheilodactylus fasciatus)*	21	**0.39**	0.68	0.13	0.00	0.50	1.13	0.13	0.25	0.50	0.50
Ariidae	White seacatfish *(Galeichthys feliceps)*	16	**0.34**	0.65	0.13	0.63	0.50	0.38	0.38	0.13	0.25	0.38
Mxyinidae	Six-gill hagfish *(Eptatretus hexatrema)*	13	**0.33**	0.78	0.00	0.00	0.00	0.00	0.50	0.50	1.13	0.50
Sparidae	Blacktails *(Diplodus sargus capensis)*	10	**0.31**	0.89	0.50	0.50	0.00	0.00	0.13	0.13	0.50	0.75
Parascorpididae	Jutjaw *(Parascorpis typus)*	15	**0.30**	0.61	0.00	0.00	0.63	0.38	0.38	0.38	0.38	0.25
Sparidae	White stumpnose *(Rhabdosargus globiceps)*	7	**0.30**	1.00	0.88	1.00	0.00	0.38	0.00	0.00	0.00	0.13
Dichistiidae	Galjoen *(Dichistius capensis)*	7	**0.22**	0.79	0.00	0.00	0.88	0.00	0.38	0.25	0.13	0.13
Sparidae	Panga *(Pterogymnus laniarus)*	5	**0.22**	1.06	0.00	0.00	0.13	1.63	0.00	0.00	0.00	0.00
Sparidae	Janbruin *(Gymnocrotaphus curvidens)*	9	**0.19**	0.53	0.00	0.00	0.50	0.63	0.00	0.13	0.25	0.00
Sparidae	Blue hottentot *(Pachymetopon aeneum)*	7	**0.17**	0.52	0.00	0.00	0.00	0.25	0.00	0.00	0.38	0.75
Sparidae	Zebras *(Diplodus cervinus hottentotus)*	5	**0.16**	0.65	0.00	0.00	0.13	0.00	0.50	0.50	0.00	0.13
Sparidae	Red stumpnose *(Chrysoblephus gibbiceps)*	5	**0.16**	0.70	0.00	0.00	0.00	0.00	0.88	0.13	0.00	0.25

Table 4: *continued*

			RELATIVE ABUNDANCE									
			ALL		ST. JAMES		BOULDERS		CASTLE ROCKS		PAULSBERG	
Family	Species	Freq.	**SITES**	SD	NT	EXP	NT	EXP	NT	EXP	NT	EXP
Oplegnathidae	Cape knifejaw *(Oplegnathus conwayi)*	5	**0.11**	0.44	0.00	0.00	0.38	0.13	0.13	0.00	0.13	0.13
Triakidae	Spotted gully-shark *(Triakis megalopterus)*	7	**0.11**	0.31	0.00	0.00	0.25	0.38	0.13	0.00	0.00	0.13
Dasyatidae	Short-tailed stingray *(Dasyatis brevicaudata)*	5	**0.08**	0.27	0.25	0.00	0.00	0.00	0.13	0.00	0.13	0.13
Tetraodontidae	Evileye pufferfish *(Amblyrhynchotes honckenii)*	4	**0.08**	0.32	0.25	0.00	0.00	0.00	0.13	0.25	0.00	0.00
Hexanchidae	Broadnose sevengill cow shark *(Notorynchus cepedianus)*	1	**0.05**	0.38	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.00
Gobiidae	Barehead goby *(Caffrogobius nudiceps)*	3	**0.05**	0.21	0.00	0.13	0.13	0.13	0.00	0.00	0.00	0.00
Cheilodactylidae	Bank steenbras *(Chirodactylus grandis)*	2	**0.03**	0.18	0.00	0.00	0.13	0.13	0.00	0.00	0.00	0.00
Sciaenidae	Dusky kob *(Argyrosomus japonicus)*	1	**0.03**	0.25	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Myliobatidae	Eagleray (*Myliobatis aquila*)	1	**0.03**	0.25	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00
Pomatomidae	Elf *(Pomatomus saltatrix)*	1	**0.02**	0.13	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Triakidae	Soupfin shark *(Galeorhinus galeus)*	1	**0.02**	0.13	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00

MaxN counts ranged from one to 105 individuals. The most abundant family were sparids (n = 1,348), followed by scyliorhinids (n = 312) and carangids (n = 283). The remaining 15 families accounted for an overall abundance of 248 individuals. The least abundant family pomatomids comprised only one individual.

Hottentot was the most abundant fish species, accounting for 21% of the overall fish abundance (Table 4). Like hottentot, the three next abundant species were shoaling species that included Cape horse mackerel (13%), strepie (12%) and steentjie (8%). Of these four most abundant species, strepie appeared in the highest numbers at any one site and had the greatest range of individuals observed from one to 105. The other shoaling species also varied considerably in number. Hottentot ranged from one to 49, Cape horse mackerel from 22 to 79 and steentjie from one to 51 individuals. Each of the remaining 32 species accounted for less than 10% of the total recorded fish abundance.

Boulders had the highest total abundance (total MaxN) of species recorded of all sites (820), followed by Castle Rocks (601), Paulsberg (500) and St. James (271). Only three species, Cape horse mackerel, hottentot and steentjie, accounted for 75% of the total abundance recorded at Boulders (Figure 7). In comparison, 75% of the total abundance at St. James was comprised of five species (strepie, hottentot, roman, steentjie and puffadder shyshark), Paulsberg of six species (hottentot, strepie, fransmadam, puffadder shyshark, roman, and pyjama catshark), and Castle Rocks of seven species (strepie, fransmadam, hottentot, roman, puffadder shyshark, Cape horse mackerel and pyjama catshark) (Figure 7).

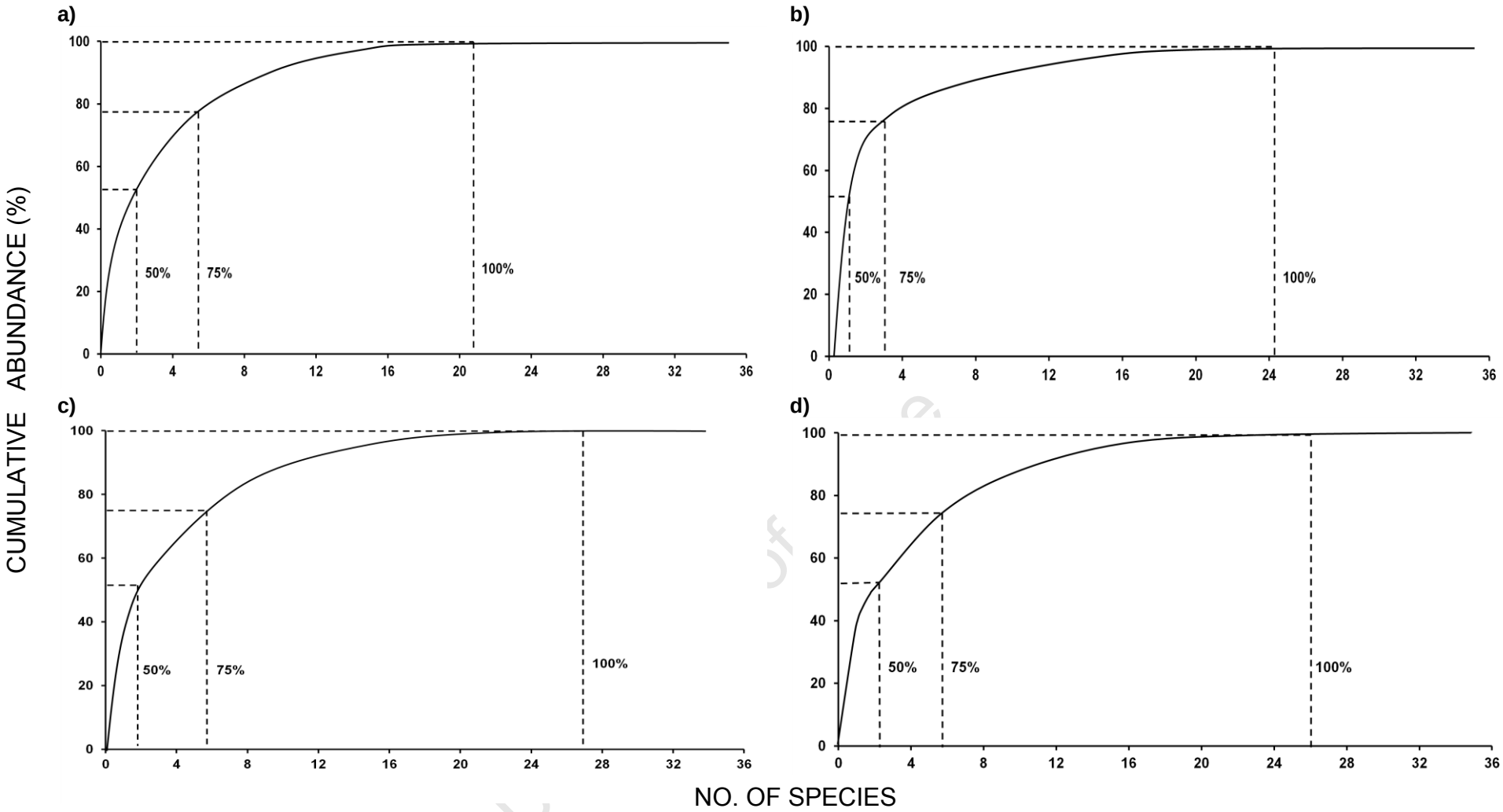

Figure 7: Cumulative abundance of species at St. James (a) Boulders (b) Castle Rocks (c) Paulsberg (d) for the no-take zones and exploited areas combined within the Table Mountain National Park Marine Protected Area based from untransformed MaxN data collected during baited remote underwater video surveys.

There was no consistent pattern between relative abundance and protection levels. None of the species was recorded at consistently higher or lower relative abundances in either the no-take zones or exploited areas (Table 4). Castle Rocks was the only site to have a higher relative abundance of species within its no-take zone (Table 4). Sixteen species in total were found to be more abundant inside than outside Castle Rocks no-take zone. These comprised four species of sparids (roman, hottentot, fransmadam and red steenbras), all four species of scyliorhinids (puffadder shyshark, pyjama catshark, dark shyshark and leopard catshark), one species of triakid (spotted gully-shark), cheilodactylid (two-tone fingerfin), hexanchid (broadnose sevengill cowshark), dasyatid (short-tailed stingray), ariid (white seacatfish) and clinid (super klipfish). Only six species were found to be more abundant in the exploited area. Of these six, three were shoaling species including Cape horse mackerel, strepie and steentjie. The remaining three species were redfingers, janbruin, and barehead goby. Four species including six-gill hagfish, blacktails, jutjaw, and zebras were found in equal relative abundance inside and outside Castle Rocks.

For Boulders, a total of 22 species differed in relative abundance between protection levels (Table 4). This was split equally between eleven species in the no-take zone and eleven species in the exploited area. Of the species in the exploited area with a higher relative abundance, seven were sparids (hottentot, steentjie, roman, white stumpnose, panga, janbruin, and blue hottentot), two were triakids (spotted gully-shark and soupfin shark) with one scyliorhinid (leopard catshark) and cheilodactylid (redfingers). For those species inside, only the zebras of the sparid family were higher in abundance. The remaining ten species were scyliorhinids (puffadder shyshark, pyjama catshark and dark shyshark), ariids (white seacatfish), dichistiids (galjoen), parascorpidids (jutjaw), cheilodactylids (two-tone fingerfin), oplegnathids (Cape knifejaw), carangids (Cape horse mackerel) and clinids (super klipfish). Bank steenbras and barehead goby were found in equal relative abundance between protection levels.

A total of 13 species were higher in relative abundance in the exploited area of Paulsberg (Table 4). Only eight species were higher in relative abundance inside the no-take zone. Three species of sparid including hottentot, strepie and janbruin had a higher relative abundance inside the no-take zone. The three other species were mxyinids (six-gill hagfish), parascorpidids (jutjaw), and clinids (super klipfish). Of the species outside, eight were sparids including roman, red steenbras, and red stumpnose. Furthermore, three species of

scyliorhinids including dark shyshark, pyjama catshark and leopard catshark were higher in abundance in the exploited area. The remaining two species formed part of the families ariid (white seacatfish) and triakid (spotted gully-shark). Five species (steentjie, redfingers, galjoen, Cape knifejaw and short-tailed stingray) were found in equal relative abundance between the no-take zone and exploited area.

St. James was found to have a total of 14 species with a higher relative abundance in its respective exploited area. Only five species were found to have a higher relative abundance inside the no-take zone of St. James. These five species inside included hottentot, redfingers, short-tailed stingray, evil-eye pufferfish, and dusky kob. All scyliorhinids (puffadder shyshark, pyjama catshark, dark shyshark and leopard catshark) were found to be more abundant outside than inside the no-take zone. In addition, five sparids (including strepie, steentjie, roman, fransmadam, and white stumpnose), and one clinid (super klipfish), cheilodactylid (two-tone fingerfin), ariid (white seacatfish), gobid (barehead goby) and myliobatid (eagleray) were recorded in higher abundance in the exploited area.

Relative abundance of four important reef fish species

There was a significant difference in the relative abundance of four important reef fish species among sites (Table 5). Apart from hottentot, the relative abundance of roman (3.688; max = 8), pyjama catshark (1.875; max = 4) and red steenbras (1.00; max = 4) was highest at Castle Rocks (no-take zone and exploited area combined) (Figure 8a-d). The relative abundance of hottentot at Boulders (13.69; max = 49), compared to Paulsberg (6.68; max = 29), Castle Rocks (5.25; max = 19) and St. James (2.56; max = 8) was more than double that found at any other site (Figure 8a).

There was no significant difference in the relative abundance of the four species between protection levels (Table 5). Only roman was found to have a significant interaction between protection levels. This can be attributed to the higher relative abundance of roman found within the exploited area of Boulders, and the higher relative abundance found in the no-take zone of Castle Rocks (Figure 8b-c).

There was no significant difference in the relative abundance of hottentot, pyjama catshark, and red steenbras among depth categories (Table 5). There was, however, a significant difference in the relative abundance of roman among depths with higher abundances over

deeper habitats (Figure 9). Hottentot, roman, and puffadder shysharks were the most abundant species at depths between 0 to 16 m, whereas steentjie and hottentot were the most abundant at depths between 16 to 32 m.

Sample sizes were unbalanced between reef profile categories to an extent that prevented meaningful statistical analysis (Table 5).

Table 5: Influence of site, protection level, depth, reef profile, and the interaction between site and protection on the abundance of four important reef fish species during baited remote underwater video surveys as indicated by generalised linear models.

Model	Distribution	Link	R^2	Δ Deviance Explained (%)	Effect	Df.	p	
Hottentot	Negative binomial	Log	0.34	80	Site	3	< 0.001	***
				0	Protection level	1	0.890	n.s.
				0	Depth	1	0.797	n.s.
				0	Profile	1	0.982	n.s.
				20	Site x Protection level	3	0.068	n.s.
Roman	Poisson	Log	0.40	27	Site	3	0.024	*
				4	Protection level	1	0.246	n.s.
				22	Depth	1	0.005	**
				4	Profile	1	0.228	n.s.
				42	Site x Protection level	3	0.002	**
Pyjama catshark	Poisson	Log	0.30	75	Site	3	< 0.001	***
				0	Protection level	1	1.00	n.s.
				2	Depth	1	0.468	n.s.
				7	Profile	1	0.128	n.s.
				16	Site x Protection level	3	0.151	n.s.
Red steenbras	Poisson	Log	0.34	80	Site	3	< 0.001	***
				0	Protection level	1	0.890	n.s.
				0	Depth	1	0.797	n.s.
				0	Profile	1	0.982	n.s.
				20	Site x Protection level	3	0.068	n.s.

*** $p < 0.001$
** $p < 0.01$
* $p < 0.05$
n.s. = not significant

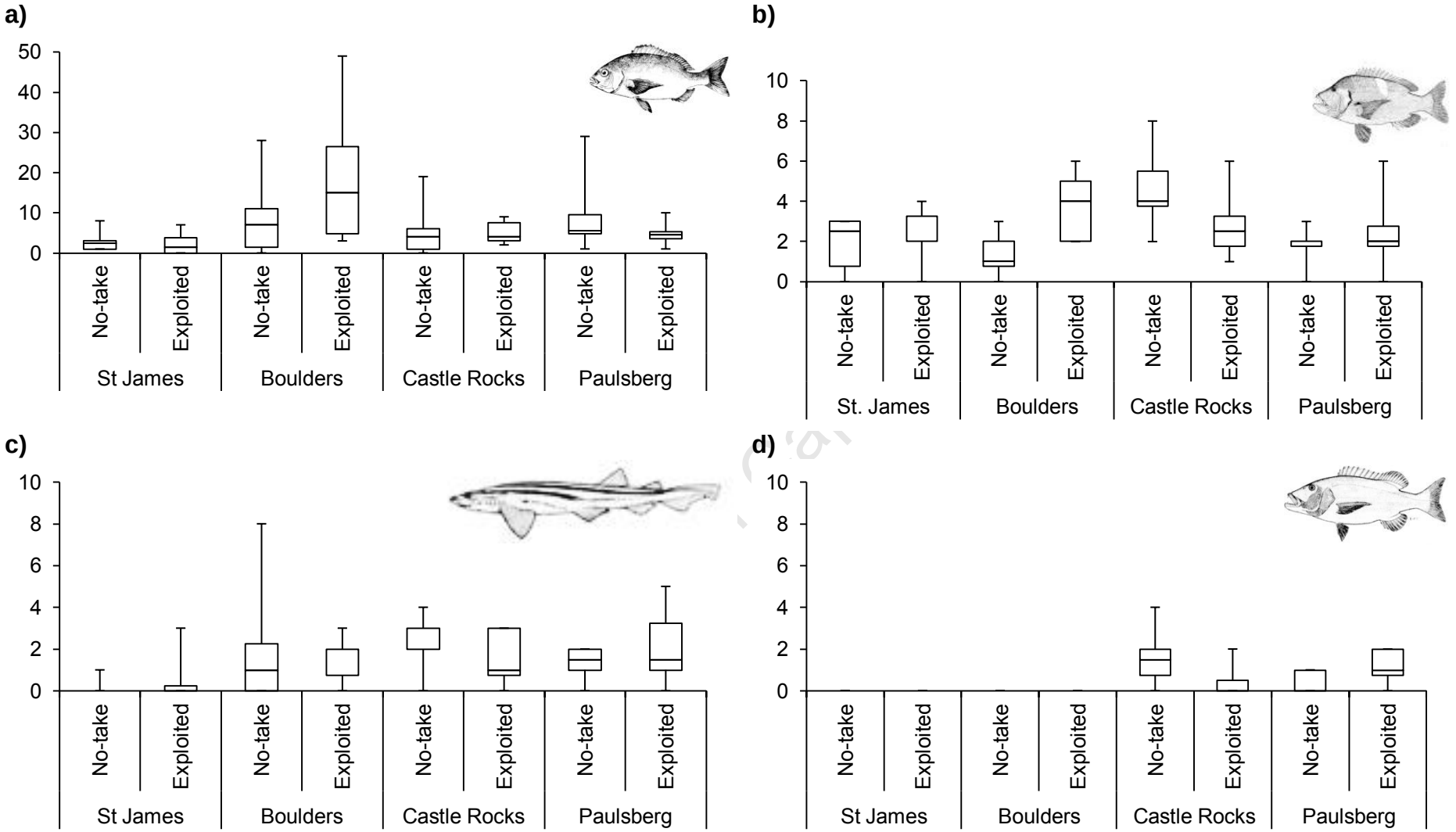

Figure 8: The relative abundance of four important reef fish species: hottentot (a) roman (b) pyjama catshark (c) red steenbras (d) between the no-take zones and exploited areas within the Table Mountain National Park Marine Protected Area recorded using baited remote underwater video surveys. The mean, first and second quartiles, and error bars representing the maximum and minimum values are shown.

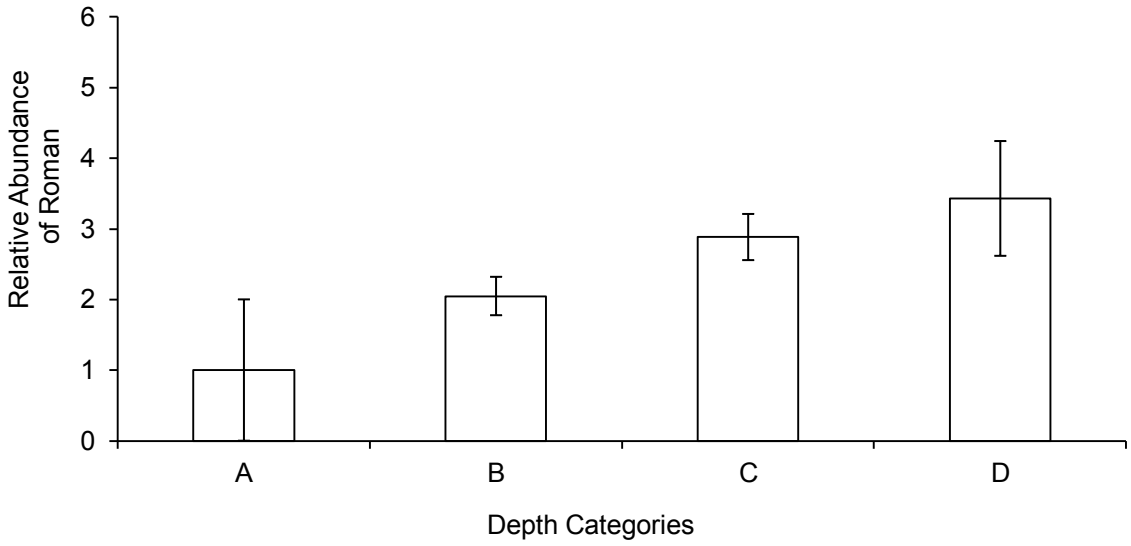

Figure 9: Relative abundance of roman recording during baited remote underwater video survey associated to each depth category (A (< 4 m), B (4 - < 8 m), C (8 - < 16 m) and D (16-32 m)). Error bars represent standard error.

Multivariate results

Community structure

One-way ANOSIM highlighted a significant difference in species composition among sites (Global R = 0.244, $p < 0.05$), except between Castle Rocks and Paulsberg. SIMPER analysis indicated that species composition between these two sites was highly similar, 41% in Castle Rocks and 45% in Paulsberg. Hottentot, roman, and puffadder shyshark cumulatively contributed over 50% to this similarity. Boulders was on average the most dissimilar with all other sites, namely, St. James (81%), then Paulsberg (72%) and Castle Rocks (72%). Hottentot, Cape horse mackerel, and steenjie all contributed over 50% to the dissimilarity between Boulders and the other three sites. As for St. James, ANOSIM results showed that it was 69% dissimilar with Paulsberg and 71% dissimilar with Castle Rocks.

PERMANOVA results also found a significant difference in species composition among sites (Table 6). Despite these differences though, the MDS plot does not show any clear groupings (Figure 10), although samples from the same sites tend to be closer together. The cluster dendrogram of the community assemblage data showed no latitudinal trend and for this reason, is not shown in the results.

Table 6: Summary of PERMANOVA analysis of patterns of reef fish assemblages (untransformed abundance [MaxN] data) in relation to site, protection level, depth, and their interactions. Because of limited sample sizes, only two factor models were run.

Metric	Source of variation	Df.	MS	SS	Pseudo-F	p	
Species Composition	Site	3	9420	28259	4.50	0.001	***
	Protection level	1	2107	2107	1.01	0.442	n.s.
	Site x Protection level	3	3144	9431	1.50	0.04	*
	Residual	56	2095	117000			
	Site	3	5550	16650	2.45	0.071	n.s.
	Depth	3	3601	10802	1.71	0.014	*
	Site x Depth	2	1835	3670	0.87	0.59	n.s.
	Residual	55	2106	116000			
	Depth	3	5539	16616	2.40	0.001	***
	Protection level	1	2104	2104	0.92	0.538	n.s.
	Depth x Protection level	2	2275	4551	0.99	0.475	n.s.
	Residual	57	2308	132000			

***p < 0.001
*p < 0.05
n.s. = not significant

There was no significant difference in species composition between protection levels (Table 6) because species were recorded in similar frequencies in both no-take zones and exploited areas (Figure 11). This can also be seen in the MDS plot where samples did not cluster according to protection levels (Figure 10). A one-way ANOSIM (site within protection levels), indicated that there were significant differences among varying sites and protection levels (*e.g.* Castle Rocks no-take zone vs St. James exploited area), but not between protection levels within sites (*e.g.* Caste Rocks no-take zone versus Castle Rocks exploited area). These differences led to a significant interaction between site and protection levels (Table 6).

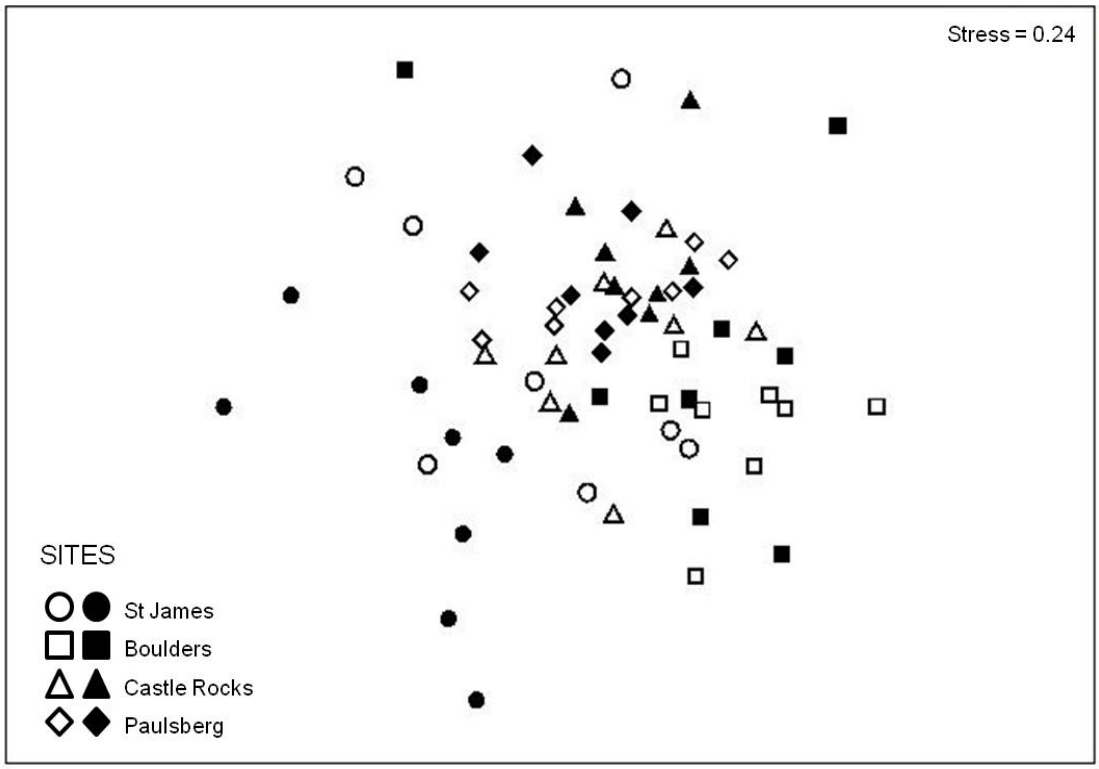

Figure 10: Multidimensional scaling (MDS) plot based on untransformed fish assemblage data. Each data point represents one baited remote underwater video deployment conducted either in the no-take zone (filled symbols) or the exploited area (clear symbols) of a site (St. James = circle, Boulders = square, Castle Rocks = triangle, Paulsberg = diamond).

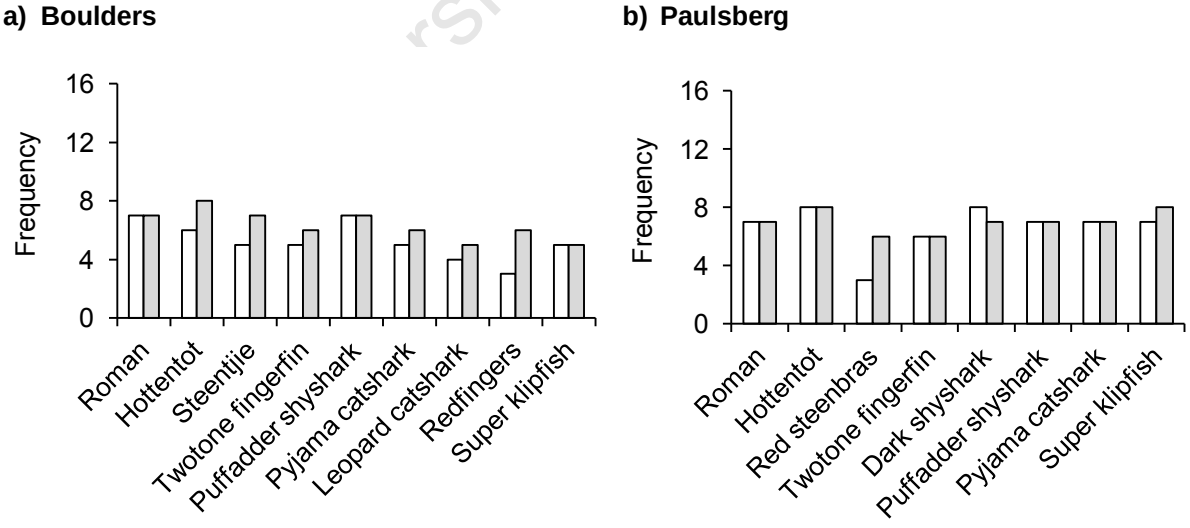

Figure 11: A subset of the data showing the frequency with which species were observed within the no-take zones (white) and exploited areas (grey) at Boulders and Paulsberg. The similarity of the frequencies of species recorded between protection levels explains why no significant differences in species composition were observed between protection levels.

One-way ANOSIM results indicate that there is a significant difference in species composition among depth categories (Global R = 0.188, $p < 0.05$). All depth categories were significantly different from each other ($p < 0.05$), except depth category B (> 4-8 m) and C (>8-16 m). SIMPER analysis indicated that species composition between these two depth categories was highly similar, 33% in depth category B and 35% in depth category C. Hottentot and roman cumulatively contributed over 50% to the similarity in depth category B and hottentot, roman and puffadder shyshark cumulatively contributed over 50% to the similarity in depth category C. Although depth categories A (< 4 m) and D (16-32 m) were the most dissimilar in species composition, there were not enough samples within depth category A to provide meaningful statistical results. However, if category A is removed from the analysis, categories B and D were the next depths most dissimilar in species composition (76%) followed by categories C and D (71%). Shoaling species including steentjie, hottentot, Cape horse mackerel and strepie all contributed over 50% to these dissimilarities among depths.

PERMANOVA results showed that there was a significant difference in species composition among depth categories (Table 6). The MDS plot also shows the difference in species composition between shallow and deep samples, however there are no clear groupings (Figure 12).

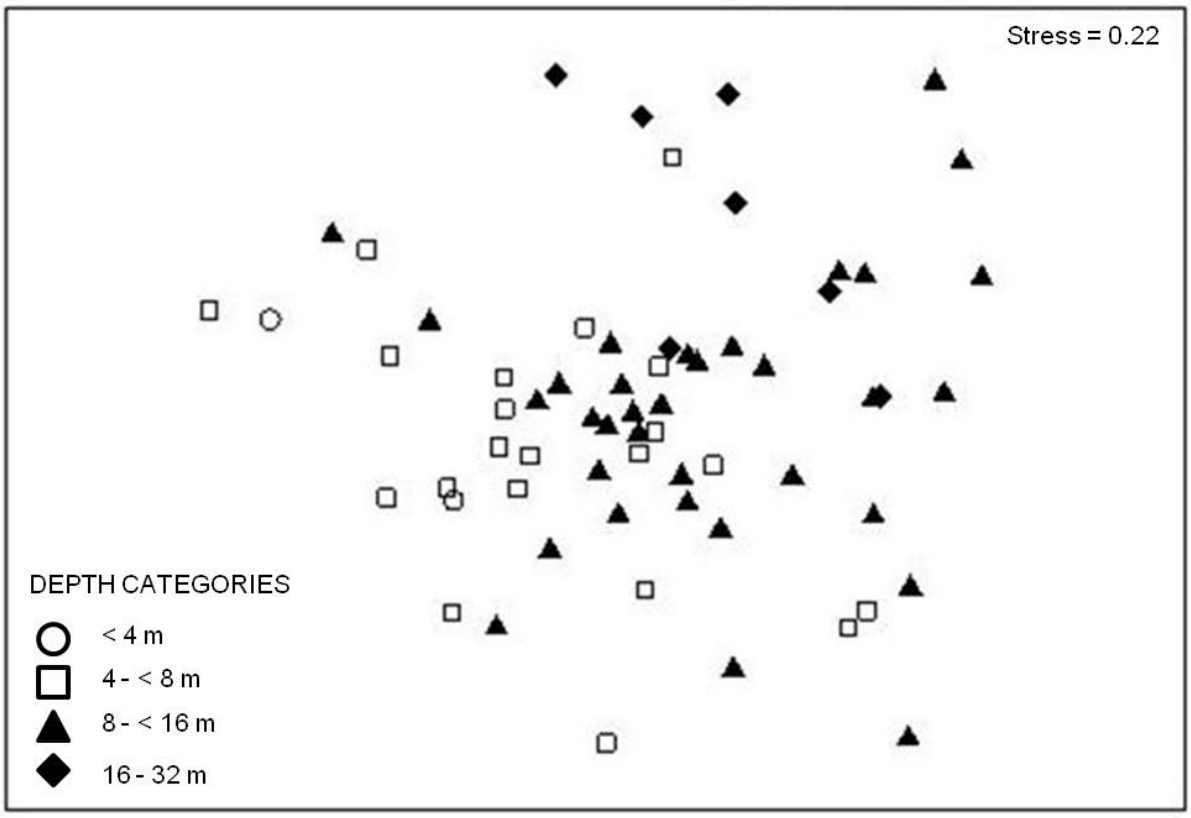

Figure 12: Multidimensional scaling plot (MDS) based on untransformed fish assemblage data. Each data point represents one of 64 baited remote underwater video deployments categorised into four depth bins, visualised by four differently shaped symbols (< 4 m = clear circle, 4 - < 8 m = clear square, 8 - <16 m = filled triangle, 16 – 32 m = filled diamond)

DISCUSSION

This study presents the first comparative assessment of the relative abundance and diversity of temperate reef fish communities between no-take zones and exploited areas, within the TMNP MPA. Results found no consistent response to protection at either the community or individual species level among the four sites. This suggests that the physical features of the sites are more powerful predictors of community structure and species abundance than their protection level.

Differences in the relative abundance and diversity of fish between protection levels were most notable around Castle Rocks. Protected reef fish assemblages at Castle Rocks diverged from the respective exploited assemblages in a number of ways, including a higher species diversity and a higher relative abundance of fish. Furthermore, Castle Rocks was the only site to frequently observe more fish in its no-take zone than in its exploited area. The differences that have been recorded within Castle Rocks no-take zone mirror earlier findings (Lechanteur 1999; Kerwath 2007; Kerwath et al. 2008).

A comparative study by Lechanteur (1999) conducted in 1994 using UVC, found that the relative reef fish abundance was also greater within the Castle Rocks no-take zone than on adjacent exploited reefs. With regards to individual species, the higher relative abundance of hottentot, roman, fransmadam, red steenbras, red stumpnose, Cape knifejaw, and broadnose sevengill cow shark in the study by Lechanteur was also observed in this study, although not all were statistically significant. Similarly, the species which did not appear to benefit from protection in 1994 are the same as those recorded during this survey. They include pelagic species (Cape horse mackerel), nomadic species (strepie, blacktail and zebras) and those not targeted by the linefishery (steentjie, redfingers and jutjaw). The presence of red steenbras and broadnose sevengill cow sharks in Castle Rocks is noteworthy because the range of these species is greater than the boundaries of this no-take zone. It is likely that red steenbras and broadnose sevengill cow sharks are attracted to certain habitat features such as the kelp channels and gullies found within Castle Rocks (Lechanteur 1999). Red steenbras are also territorial as juveniles, only migrating when adult, and so may find sufficient refuge within this no-take zone (Brouwer et al. 2002). Although fish size could not be accurately measured, several juvenile red steenbras, characterised by their yellow belly and tail, were observed in the BRUV footage (*pers. obs.*). Overall, for residential and territorial species, the

resemblance in the relative abundances and diversity of fish species between the two studies, over time, suggests that the area protected by Castle Rocks is of sufficient extent to largely or even fully contain the home ranges of many species.

Based on area utilisation and activity patterns of roman, which are known to occupy small home ranges of between 1000 m^2 and 3000 m^2, Castle Rocks is of sufficient size (3 km^2) and suitable habitat structure to provide protection from exploitation (Kerwath et al. 2007). The failure of roman to display a density gradient across the reserve boundary at Paulsberg, a site similar in size, depth, and habitat structure to Castle Rocks (C.G. Attwood, UCT, *pers. comm.*), may indicate that the time elapsed since a site was placed under protection is an explanatory factor.

An individual based model for roman developed by Kerwath et al. (2008) suggested that populations would only return to their pre-exploitation levels following ten years of protection. This estimate was based on recovery within Castle Rocks, a site which has been under varying protection since 1979. Such a slow recovery rate corroborates results from other studies on reef fish species (Buxton 1993; Russ & Alcala 2004) and is presumably due to the slow growth rate, and the high exploitation levels, which reef fish have experienced (Halpern & Warner 2002; Barrett et al. 2007). Despite this difference, the species abundance, diversity and species composition between Castle Rocks and Paulsberg were highly similar and the presence of roman, red stumpnose, and red steenbras, in particular, indicates that these sites provide the most suitable habitat for sparid species with collapsed or over-exploited populations.

Commercially-important shoaling species such as hottentot, Cape horse mackerel, steentjie and panga, exhibited a preference for deeper water habitats (16-32 m). As it extends the furthest into the bay and is the deepest of all sites, Boulders consequently accumulated the highest relative abundance of fish as a result of large aggregating shoals of the aforementioned species. Although there was no significant difference in depths between the no-take zones and exploited areas at Boulders, the exploited areas were deeper on average and, for this reason, likely accumulated a higher relative abundance than inside the no-take zone. The higher abundance of fish recorded at Boulders is likely due in part to the depth refuge it may provide from exploitation (Goetze et al. 2011). Unlike the other no-take zones in False Bay where spearfishermen ‘fish the line’ with the hopes of catching larger and bigger fish (S. Benjamin, Animal Ocean, *pers. comm.*), they are restricted at Boulders as they can

only dive at depths shallower than 15 m (C.G. Attwood, UCT, *pers. comm.*). By being less accessible to exploitation, Boulders may be less impacted by fishing activities than the other sites.

The relative abundance of roman was also found to increase significantly with depth. This relationship with depth differs from other studies, which have found roman to be more abundant at shallower sites (Buxton & Smale 1989; Götz et al. 2009). However, the difference may be explained by the fact that roman prefer high profile reef for the food, shelter, and low turbidity conditions it provides (Buxton & Smale 1989; Friedlander et al. 2003; Lechanteur 1999). Unlike in the Tsitsikamma and Goukamma MPAs, where shallow sites are characterised by high profile reef (Buxton & Smale 1989; Götz et al. 2009), the shallow sites of False Bay, such as St. James, are underlain by Malmesbury shale favouring the formation of low profile reef (Spargo 1991; Compton 2004). High profile reef is only present at Boulders, Castle Rocks, and Paulsberg, where the geology of these deeper sites is characterised by large granite boulders (Spargo 1991; Compton 2004). The geological differences together with the refuge provided by depth may provide the best explanation for the patterns observed in the distribution and abundance of roman.

Species composition at St. James differed substantially from the other three sites. St. James had the fewest recorded species, and was the only site in which dusky kob, an inshore species of turbid waters, and white stumpnose, an estuarine-dependent species, were observed (Day 1970; Clark et al. 1994). Whilst the abundance of hottentot, redfingers, short-tailed stingrays, evil-eye pufferfish, and dusky kob was greater within the St. James no-take zone compared to the exploited area, the relative abundance of the other nine species recorded at this site was not. These results suggest that St. James provides the least suitable habitat for reef fish species of all sites. This is, however, a factor inherent to the original design objectives of St. James (C.G. Attwood, UCT, *pers. comm.*).

The no-take zones along the western side of False Bay were not intended to be ecological replicates, but rather were created to complement one another in terms of the variety of species and habitats they protected (Hockey & Branch 1997). They were promulgated to ensure representivity of a number of features, processes, and ecological services within a network of no-take zones (Hockey & Branch 1997). Only two of the sites, namely Castle Rocks and Paulsberg, were proclaimed with reef fish protection as an objective (C.G. Attwood, UCT, *pers. comm.*). This is reflected in the results provided from this study. The

Boulders no-take zone on the other hand, was designed to protect a nesting colony of the endangered African penguin (*Spheniscus demersus*), whilst the St. James no-take zone was created to provide an undisturbed shoreline for the field based education of school children (C.G. Attwood, UCT, *pers. comm.*). The failure to detect a higher abundance and diversity of reef fish species within St. James is a function of its small size (0.3 km^2) and the fact that it contains mostly low profile, less suitable reef habitat (C.G. Attwood, UCT, *pers. comm.*). Furthermore, if there is a protective effect within a small no-take zone, finding it is difficult, as the benefits of protection will accrue more slowly in small no-take zones, as fish are less likely to encounter them and less likely to stay in them (Halpern & Warner 2002).

The spatial variability in the abundance and diversity of fish observed across sites can also be explained by the position of each no-take zone within the bay. Lying on a continuum from the northern shore to the opening of the bay, the four no-take zones are each subject to different physical attributes, such as water circulation, turbidity and wind speed (Clark et al. 1996). Ocean current patterns within False Bay are complex and cannot be explained by wind or tide alone (Wainman et al. 1987). False Bay is not narrow enough to act like an estuary and neither is it exposed enough for deep-sea conditions to have a predominant effect on its currents and water temperatures (Grundlingh & Largier 1991). Although current reversals are frequent, water circulation is generally anticlockwise (Wainman et al. 1987). Rich organic material, lifted by wave action, circulates from the middle of the bay in a westerly direction along the northern shores (Clark et al. 1996). Based on these currents and the freshwater discharge from the estuaries that enter the bay, water turbidity is highest in the north of the bay and lowest towards the south at the peninsula (Clark et al. 1996). Of the four sites, water is most turbid at St. James but becomes progressively clearer towards Boulders, Castle Rocks, and Paulsberg. Although this study did not quantify water turbidity, a profound difference in water clarity was observed among the sites that could explain the differences recorded during BRUV surveys. Conditions of low turbidity and high visibility were also found to favour a higher abundance of sparid species during controlled angling and UVC in the Goukamma MPA, mirroring the observations made during these BRUV surveys (Götz et al. 2009).

The water circulation within the bay may also be responsible for the similarity in species assemblages and diversity observed at Castle Rocks and Paulsberg. Water, flowing anticlockwise, has the potential to transport larvae from broadcast spawners in Castle Rocks

to Paulsberg (Teske et al. 2010). In addition, being closer together than any other sites, at only a distance of 1 km apart, there is a greater chance for fish from Castle Rocks such as roman to 'spill-over' into Paulsberg (Kerwath et al. 2008). Much like Tobler's law, which explains that 'near things are more related than distant things' (Tobler 1970, p.3), Castle Rocks and Paulsberg are likely to have similar physical conditions which may determine the similar distribution of species.

With the exception of Castle Rocks, the differences in fish abundance and diversity within sites, suggests that the results are not attributable to protection effects. This finding may be due in part to insufficient statistical power to detect population changes for the rarer species (Barrett et al. 2007). Another plausible explanation is that fishing pressure outside the no-take zones is not intensive enough to affect reef fish assemblages, as explained by the intermediate disturbance hypothesis (Chapter 1), with the notable exception of roman and red steenbras at the time when the no-take zones were declared. Since the emergency declaration in 2000 and the reduction in commercial linefishing effort (as stipulated in the MLRA, Section 16), there has been a reduced demand for overexploited reef fish (C.G. Attwood, UCT, *pers. comm.*). This, coupled with a severe reduction in bag limits, has made it less economically viable for fishermen to continue to target reef species (C.G. Attwood, UCT, *pers. comm.*). In response, the linefishery has turned its efforts towards snoek (*Thrysites atun*) and yellowtail (*Seriola lalandi*), the migratory pelagic species. It is probable that this shift has lowered fishing intensity to such an extent that the fishing differential is not strong enough for protection to show a response (Barrett et al. 2007).

The differences between fish abundance and diversity were more evident in Castle Rocks, indicating that prior to its promulgation, fishing pressure had caused an alteration in the species composition. Castle Rocks is located next to Miller's Point, a popular slipway, so it is possible that it was subjected to higher fishing pressure prior to its protection than the other three sites. The other no-take zones, although also near to popular fishing locations, were designated because they were historically the least disturbed and least fished areas in the bay (Clark et al. 1996).

Poaching within the no-take zones remains a possibility for why no single unifying pattern of abundance or diversity measures was evident following protection. During this study, two boats were observed fishing illegally within Boulders and one boat within Castle Rocks.

There were also numerous occasions in which boats were observed 'fishing the line' at Castle Rocks and Paulsberg.

BRUV survey methodology

The accuracy of a single method to monitor diverse fish assemblages can be highly variable (Stobart et al. 2007). As species are not all equally observable (Colton & Swearer 2010), it is important to consider the validity of the results, especially when undertaking an assessment on the abundance and diversity of fish between no-take zones and exploited areas.

Baited remote underwater video was an effective tool for monitoring a diverse assemblage of species in the TMNP MPA. A wide range of families was attracted to the bait, indicating that BRUV is a feasible method with which to document the presence or absence of species. When compared to controlled angling, UVC and other BRUV surveys in the Goukamma, Tsitsikamma, and Stilbai MPAs respectively, the diversity of fish recorded during this study (36 species representing 18 families) was similar (Bennett et al. 2009; Götz et al. 2009; Bernard & Götz 2012; De Vos 2012). In a previous study at Castle Rocks in particular, Lechanteur (1999) using UVC, recorded 28 species and 11 families of fish during 795 point counts. In comparison, this thesis recorded 27 species and 14 families of fish at this site during 64 BRUV deployments.

Baited remote underwater video surveys are ideally suited to monitoring within no-take zones as they are non-extractive and provide data on species targeted by fisheries (Cappo et al. 2004; Stobart et al. 2007). Of all the species recorded during this study, 14 were considered depleted and/or vulnerable to collapse according to the South Africa Sustainable Seafood Initiative (SASSI) (Appendix Table A2). Eight of these formed part of South Africa's most dominant endemic population, the sparids. Owing to its design, BRUV offers a solution to the observational biases of UVC surveys, as well as reducing the variable catchability and size selectivity issues that are inherent to angling techniques (Willis et al. 2000; Cappo et al. 2004; Stobart et al. 2007).

However, despite its advantages, the abundance values from over 64 hours of video footage was still too low for statistical analysis of data on the rare species, such as red steenbras, and nomadic species, such as galjoen. Baited remote underwater video was also limited by its ability to record shoaling species as the number of fish that could be seen within the field of

view was limited (Willis et al. 2000; Stobart et al. 2007; Bernard & Götz 2012). Furthermore, the presence of sharks and the antagonistic behaviour of roman were frequently observed to cause a competitive exclusion of other species around the bait. These limitations have reduced the strength of the conclusions from this study.

A widespread effect of exploitation, which was not considered in this study, was a decrease in fish body size. There are many comparative studies which have found no difference in fish abundances during comparisons, but have instead recorded a significant decrease in the body size of fish in exploited areas, in particular of target species (Côté et al. 2001; Barrett et al. 2007). Broadcast spawners especially, whose recruitment strength is similar across no-take zones and exploited areas, are more likely to experience a truncated size distribution than a change in abundance or diversity. If these changes go unnoticed, life-history characteristics, migration patterns, predator-prey relationships and even competitive interactions may potentially be altered (Attwood et al. 1997; Shin et al. 2005). Single BRUV cameras are however, unable to accurately measure fish.

Conclusion

This study has revealed that there are considerable differences in reef fish assemblages even over small spatial scales. Whilst the differences observed among sites in this study appear to be driven by physical parameters, the relative abundance of individual species is likely influenced by a combination of different physical and biological effects that operate simultaneously. Single BRUV camera systems have shown that they are a useful method to simultaneously assess the relative abundance and diversity of fish. However, their inability to accurately measure fish size is a severe disadvantage (Bernard & Götz 2012). A single BRUV camera will therefore prove useful to detect recruitment failure or conversely recruitment success, but, by itself, should be considered insufficient to assess the effectiveness of no-take zones.

CHAPTER 3

Study Review and Synthesis

CONCLUSIONS

Conservation Implications

This study determined that variation exists in the composition and abundance of reef fish communities among the four no-take zones within the Table Mountain National Park (TMNP) Marine Protected Area (MPA). These findings indicate that for comparability, future assessments within the TMNP MPA should be designed to target similar locations and depth ranges within the bay.

Castle Rocks and Paulsberg, the two no-take zones designed specifically for the purpose of reef fish conservation, supported the highest species diversity. This, coupled with the detection of rare species, such as red stumpnose and red steenbras, inside and outside their boundaries, is an encouraging sign for fisheries management. Although Boulders and St. James were proclaimed for different reasons, they too harboured important populations of reef fish. Owing to its depth, Boulders recorded the highest abundance of reef fish, whilst St. James was the only site to observe estuarine-dependent species. Based on their frequency of occurrence, multiple commercially-important species including hottentot and roman appear to be successfully recruiting both inside and outside the no-take zones. It is evident from these results that although these small no-take serve different purposes, they are each valuable to the TMNP MPA and contribute to achieving its conservation and fisheries management goals.

Technical suggestions

As the first study to implement the new baited remote underwater video (BRUV) system using GoPro® HERO 2 HD camera (Woodman Labs 2009) and a mild steel rig in South Africa, it is encouraging to note that, thanks to its design, it successfully landed on reef without being dragged by the currents. Furthermore, the use of the chain weight ensured the camera landed horizontally for the majority of deployments.

There remain a few technical aspects which should be refined in order to improve the scientific methodology. Firstly, as visibility may influence the activity rates of fish (Buxton & Smale 1989), a measure of horizontal visibility on the bottom of the seafloor needs to be developed. The most cost-effective solution would be to attach two parallel laser pointers to the BRUV system, aimed at visible targets at varying distances along the rig.

A more accurate measure of reef profile is required than the method adopted in this study. Unlike the BRUV system used by De Vos (2012), which was 1 m above the ground, the new GoPro® HERO 2 HD camera (Woodman Labs 2009) system lies only 40 cm above the seafloor. At such an angle, reef profile appears disproportionately large. Ideally, accurate bathymetry data would solve this issue; however these data are not always available. One alternative could be to take four depth measurements in a radius of 20 m from where the BRUV system is deployed. Similar to reef profile measurements from underwater visual censuses (UVC) that use point counts, reef profile could be classified as the difference between the shallowest and deepest depth measurement (*sensu* Götz et al. 2009).

Future research

Baseline data has now been collected for the first time on the abundance and diversity of reef fish along the west side of False Bay. A long-term monitoring programme can, and should now, be established. The low costs and manpower associated with this BRUV system render much more feasible the future monitoring of South Africa's marine protected areas. With only four BRUV systems, it is possible to obtain a sample size of 20 deployments in one day (*pers. obs.*). If this was implemented once a month, a sample size of 240 deployments could be achieved on an annual timescale, greatly improving the ability to detect significant changes in the abundance and diversity of rare and target species.

Future comparisons inside and outside no-take zones must consider using stereo-cameras to collect accurate fish size measurements. A change in the size structure of exploited communities is the most noticeable effect of fishing, and therefore the best indicator to use when monitoring the effectiveness of MPAs (Shin et al. 2005; Götz et al. 2009).

A measure of fishing intensity within TMNP MPA and how it changes over time and space must be obtained in order to strengthen the results in future comparisons. Spatial information

on recreational and commercial fishing effort will help elucidate the actual effects of protection (Willis et al. 2000; Molloy et al. 2009).

In addition, the influence of habitat variability on the distribution and abundance of reef fish should be assessed within each no take zone, to distinguish the effects of protection from other sources of heterogeneity (García-Charton et al. 2000). This can be achieved by conducting BRUV deployments equitably across multiple ranges of reef profile and depth

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APPENDIX

Table A1: Environmental variables measured during baited remote underwater video surveys at four sites in the Table Mountain National Park Marine Protected Area over the four-month study period.

Month	Site	Protection level	Water temperature (°C)	Depth (m)	Reef profile
SEPTEMBER	St. James	No-take	15.23	4.6	High
		No-take	15.04	7.0	High
		Exploited	14.85	5.0	High
		Exploited	14.84	4.8	High
	Boulders	No-take	14.72	16.5	High
		No-take	14.74	17.7	Low
		Exploited	14.50	22.0	High
		Exploited	14.77	20.9	High
	Castle Rocks	No-take	15.13	9.4	High
		No-take	15.09	11.9	High
		Exploited	14.98	12.8	High
		Exploited	15.00	9.4	High
	Paulsberg	No-take	15.39	9.4	High
		No-take	15.37	9.4	High
		Exploited	15.20	7.9	High
		Exploited	15.12	6.1	High
Mean			*15.00*	*10.9*	
OCTOBER	St. James	No-take	16.43	4.7	Low
		No-take	16.49	2.6	Low
		Exploited	16.03	11.5	Low
		Exploited	16.07	7.7	Low
	Boulders	No-take	16.06	12.4	Low
		No-take	16.03	14.7	High
		Exploited	15.85	15.0	High
		Exploited	15.83	16.3	Low
	Castle Rocks	No-take	15.10	10.7	High
		No-take	15.13	8.5	Low
		Exploited	15.20	11.9	Low
		Exploited	15.18	7.1	Low
	Paulsberg	No-take	14.64	11.6	High
		No-take	14.50	13.1	High
		Exploited	14.78	4.7	High
		Exploited	14.79	5.9	High
Mean			*15.51*	*9.9*	

Table A1: *continued*

Month	Site	Protection level	Water temperature (°C)	Depth (m)	Reef profile
NOVEMBER	St. James	No-take	18.18	5.3	Low
		No-take	18.19	5.4	High
		Exploited	18.07	7.3	High
		Exploited	18.04	8.0	High
	Boulders	No-take	17.22	13.6	High
		No-take	16.80	15.7	High
		Exploited	16.61	15.2	High
		Exploited	16.51	18.6	High
	Castle Rocks	No-take	14.51	9.9	High
		No-take	15.24	11.5	High
		Exploited	15.12	10.9	High
		Exploited	15.61	10.1	High
	Paulsberg	No-take	14.70	9.5	High
		No-take	14.76	10.1	High
		Exploited	15.26	9.0	High
		Exploited	15.93	5.5	High
Mean			*16.30*	*10.4*	
DECEMBER	St. James	No-take	16.16	3.4	Low
		No-take	16.15	4.3	Low
		Exploited	16.28	5.6	High
		Exploited	16.00	6.9	High
	Boulders	No-take	15.48	15.0	High
		No-take	15.55	15.2	High
		Exploited	15.75	19.5	Low
		Exploited	16.41	14.1	Low
	Castle Rocks	No-take	16.81	15.5	High
		No-take	16.89	13.7	High
		Exploited	17.36	11.4	High
		Exploited	16.74	12.1	Low
	Paulsberg	No-take	17.75	10.0	Low
		No-take	17.91	7.0	High
		Exploited	18.21	7.3	Low
		Exploited	18.12	6.6	High
Mean			*16.72*	*10.5*	

Table A2: Species recorded using baited remote underwater video surveys in the Table Mountain National Park Marine Protected Area. Species names in bold represent linefish species assessed by the Southern African Sustainable Seafood Initiative (SASSI). These species are classified as either 'red' or 'orange' under the classification scheme. They represent species that are currently depleted and whose populations are vulnerable to collapse.

Class	Family	Species	Common Name
Myxini	Mxyinidae	*Eptatretus hexatrema*	Six-gill hagfish
Chondrichthyes	Hexanchidae	*Notorynchus cepedianus*	Broadnose sevengill cow shark
Chondrichthyes	**Triakidae**	***Galeorhinus galeus***	**Soupfin shark**
Chondrichthyes	**Triakidae**	***Triakis megalopterus***	**Spotted gully-shark**
Chondrichthyes	Scyliorhinidae	*Haploblepharus edwardsii*	Puffadder shyshark
Chondrichthyes	Scyliorhinidae	*Haploblepharus pictus*	Dark shyshark
Chondrichthyes	Scyliorhinidae	*Poroderma pantherinum*	Leopard catshark
Chondrichthyes	**Scyliorhinidae**	***Poroderma africanum***	**Pyjama catshark**
Chondrichthyes	Myliobatidae	*Myliobatis aquila*	Eagleray
Chondrichthyes	Dasyatidae	*Dasyatis brevicaudata*	Short-tailed stingray
Actinopterygii	Ariidae	*Galeichthys feliceps*	White seacatfish
Actinopterygii	Pomatomidae	*Pomatomus saltatrix*	Elf
Actinopterygii	Sparidae	*Boopsoidea inornata*	Fransmadam
Actinopterygii	**Sparidae**	***Chrysoblephus gibbiceps***	**Red stumpnose**
Actinopterygii	**Sparidae**	***Chrysoblephus laticeps***	**Roman**
Actinopterygii	**Sparidae**	***Diplodus cervinus hottentotus***	**Zebras**
Actinopterygii	**Sparidae**	***Diplodus capensis***	**Blacktails**
Actinopterygii	**Sparidae**	***Gymnocrotaphus curvidens***	**Janbruin**
Actinopterygii	Sparidae	*Pachymetopon aeneum*	Blue hottentot
Actinopterygii	**Sparidae**	***Pachymetopon blochii***	**Hottentot**
Actinopterygii	**Sparidae**	***Petrus rupestris***	**Red steenbras**
Actinopterygii	Sparidae	*Pterogymnus laniarius*	Panga
Actinopterygii	**Sparidae**	***Rhabdosargus globiceps***	**White stumpnose**
Actinopterygii	Sparidae	*Sarpa salpa*	Strepie
Actinopterygii	Sparidae	*Spondyliosoma emarginatum*	Steentjie
Actinopterygii	**Dichistiidae**	***Dichistius capensis***	**Galjoen**
Actinopterygii	Parascorpididae	*Parascorpis typus*	Jutjaw
Actinopterygii	Cheilodactylidae	*Cheilodactylus fasciatus*	Redfingers
Actinopterygii	Cheilodactylidae	*Chirodactylus brachydactylus*	Twotone fingerfin
Actinopterygii	Cheilodactylidae	*Chirodactylus grandis*	Bank steenbras
Actinopterygii	**Sciaenidae**	***Argyrosomus japonicus***	**Dusky kob**
Actinopterygii	**Oplegnathidae**	***Oplegnathus conwayi***	**Cape knifejaw**
Actinopterygii	Carangidae	*Trachurus capensis*	Cape horse mackerel
Actinopterygii	Clinidae	*Clinus superciliosus*	Super klipfish
Actinopterygii	Gobiidae	*Caffrogobius nudiceps*	Barehead goby
Actinopterygii	Tetraodontidae	*Amblyrhynchotes honckenii*	Evileye pufferfish

Table A3: Shannon-Wiener diversity index for each deployment per site.

Month	Site	Protection level	H'(loge)
SEPTEMBER	St. James	No-take	1.33
		No-take	1.28
		Exploited	1.49
		Exploited	1.33
	Boulders	No-take	1.79
		No-take	1.53
		Exploited	1.39
		Exploited	1.32
	Castle Rocks	No-take	2.30
		No-take	1.25
		Exploited	2.20
		Exploited	0.93
	Paulsberg	No-take	2.07
		No-take	2.04
		Exploited	2.46
		Exploited	2.42
OCTOBER	St. James	No-take	1.10
		No-take	1.39
		Exploited	1.47
		Exploited	1.00
	Boulders	No-take	0.74
		No-take	2.33
		Exploited	1.71
		Exploited	0.98
	Castle Rocks	No-take	2.34
		No-take	2.09
		Exploited	2.31
		Exploited	1.85
	Paulsberg	No-take	1.54
		No-take	1.71
		Exploited	1.67
		Exploited	1.59
NOVEMBER	St. James	No-take	1.72
		No-take	1.71
		Exploited	1.39
		Exploited	2.11
	Boulders	No-take	1.17
		No-take	1.04
		Exploited	1.29
		Exploited	2.04
	Castle Rocks	No-take	1.61
		No-take	2.40
		Exploited	1.54
		Exploited	1.51
	Paulsberg	No-take	1.91
		No-take	2.16
		Exploited	2.38
		Exploited	2.02

Table A3: *continued*

Month	Site	Protection level	H'(loge)
DECEMBER	St. James	No-take	1.01
		No-take	1.51
		Exploited	1.49
		Exploited	1.72
	Boulders	No-take	1.31
		No-take	0.76
		Exploited	1.80
		Exploited	1.27
	Castle Rocks	No-take	2.49
		No-take	2.07
		Exploited	1.90
		Exploited	1.97
	Paulsberg	No-take	1.91
		No-take	2.43
		Exploited	1.80
		Exploited	1.89